Recommendations of the Expert Panel to Define BMP Effectiveness for Urban Tree Canopy

Karen Cappiella, Sally Claggett, Keith Cline, Susan Day, Michael Galvin, Peter MacDonagh, Jessica Sanders, Thomas Whitlow, Qingfu Xiao

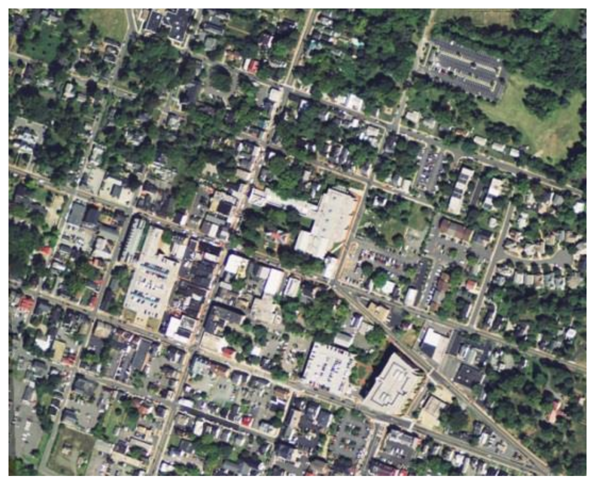

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Table of Contents

LIST OF TABLES

LIST OF FIGURES

Executive Summary

The Forestry Work Group convened an Expert Panel to determine pollution control performance estimates for the best management practice of expanded urban tree canopy as part of the Phase 6 Chesapeake Bay Watershed Model (CBWM). The Expert Panel recommendations are based on review and synthesis of the literature, best professional judgement and the approved tree canopy land use loading rates for nitrogen, phosphorus and sediment. The recommendations include a revised urban tree canopy BMP credit for tree planting. Panel members also strongly recommend the Chesapeake Bay Program convene a future expert panel for a new BMP to credit practices that maintain and conserve existing tree canopy in urban areas given the continued net loss of tree canopy throughout the Bay watershed. There was strong agreement among the Panel that the conservation and maintenance of existing tree canopy in developed or developing areas has the potential to have a greater positive impact on local efforts to address water quality than tree planting alone.

Nutrient and sediment reductions for the proposed Phase 6 urban tree canopy BMP is based on the estimated acres of tree canopy resulting from tree planting that will be simulated as a land use change in the Phase 6 model. The Expert Panel agreed to use a robust modeling approach to estimate the annual projected canopy area for new trees planted based on the number of factors that affect an individual tree's growth summarized in the literature. The creditable area for each tree planted is based on outcomes from the i-Tree Forecast model using scenarios and input parameters defined by the Expert Panel. The model scenarios explicitly account for a variety of tree species, mortality and growing conditions. Each tree planted in developed areas is eligible for a creditable area of 144 ft^2 that translates to 300 trees per acre with the average nutrient and sediment reductions summarized in Table E.1. This creditable area is based on an estimated annual growth for a 10-year old tree after planting (assuming an initial DBH of 1-inch at planting). This timeframe for growth was selected based on the timeframe for the Bay TMDL for practices "to be in the ground" and growth period for a newly planted tree to achieve sufficient size to be mapped as a land use, as it is updated. The relative land use loading rate (percent reduction) would then be applied in the Chesapeake Bay Program modeling tools as a land use change equal to the reported acreage of tree planting. The total reduction would be automatically calculated through the modeling tools, but can be understood by the following equation:

Tree Canopy Acreage from Number of Trees Planted x % loading rate reduction x underlying land use loading rate
= estimated Lbs reduced/yr (edge of field)

Table E.1. Tree canopy relative land use loading rates based on the underlying land use land cover (Source: Hynicka and Divers 2016)

Land Use	Total Nitrogen Reduction (%)	Total Phosphorus Reduction (%)	Total Sediment Reduction (%)
Canopy over Turfgrass	23.8	23.8	5.8
Canopy over Impervious	8.5	11.0	7.0

The Panel could not identify any unintended consequences of adopting the recommendations for this UTC BMP. While the Expert Panel acknowledged the contribution of nutrients in leaf litter to surface waters, the Panel members concur with the review of the Street Sweeping and Inlet Cleaning Expert Panel report that a more explicit accounting of this source requires further evaluation given the uptake and storage of nutrients from trees. Further, the Panel provides recommendations for decision rules to avoid double-counting mapped acres of existing tree canopy and acres from newly planted trees reported under the recommended annual BMP. The Expert Panel provides recommendations to advance research and management that address the current limitations of research to comprehensively quantify the effects of urban tree canopy on water quality, other than forests and riparian buffers, along with the need to focus strategies that maintain and conserve existing urban tree canopy.

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SECTION 1: CHARGE AND MEMBERSHIP OF THE EXPERT PANEL

1.1 Panel Membership

The Forestry Workgroup approved the membership of the expert panel in February 2015. The roster for the Urban Tree Canopy BMP Expert Panel is provided in Table 1. A copy of the meeting minutes is provided in Appendix A.

Table 1. Expert Panel Membership.

Name	Affiliation
Panel Members	
Karen Cappiella	Center for Watershed Protection
Sally Claggett	US Forest Service, CBPO
Keith Cline	Fairfax County (VA)
Susan Day*	Virginia Tech
Michael Galvin	SavATree
Peter MacDonagh	Kestrel Design Group
Jessica Sanders	Casey Trees
Thomas Whitlow	Cornell University
Qingfu Xiao	University of California-Davis
Panel Support	
Neely Law (Chair)	Center for Watershed Protection
Jeremy Hanson (Coordinator)	Virginia Tech, CBPO
Brian Benham	Virginia Tech (Project Director)
Marcia Fox	DE DNREC (WTWG rep)
Ken Hendrickson	EPA Region 3 (Regulatory Support)
David Wood	CRC, CBPO (CBP modeling team rep)

* On sabbatical leave starting in September 2015

1.2 Panel Charge

The Expert Panel was convened by the Forestry Work Group (FWG) of the Chesapeake Bay Program's (CBP) Water Quality Goal Implementation Team (WQGIT) to determine pollution control performance estimates for the best management practice (BMP) of expanded urban tree canopy (UTC). A literature review and synthesis was completed for the purpose of informing recommendations for this BMP. Best professional judgement and use of models provided additional insight to inform recommendations where gaps in information or understanding existed.

The Expert Panel report also includes the following:

- Identity and expertise of Panel members
- Practice name/title
- Detailed definition of the practice
- Recommended nitrogen, phosphorus, and sediment loading or effectiveness estimates
- Justification for the selected effectiveness estimates, including a list of references used and a detailed discussion of how each reference was considered, or if another source was investigated, but not considered.
- Description of how best professional judgment was used, if applicable
- Land uses to which the BMP is applied
- Load sources the BMP will address and potential interactions with other practices
- Description of pre-BMP and post-BMP circumstances, including the baseline conditions for individual practices
- Conditions under which the BMP works/does not work/or varies in its effectiveness
- Temporal performance of the BMP including lag times between establishment and full functioning (if applicable)
- Unit of measure (e.g., feet, acres)
- Locations within the Chesapeake Bay watershed where this practice is applicable
- Useful life; effectiveness of practice over time
- Cumulative or annual practice
- Description of how the BMP will be tracked, reported, and verified
- Suggestion for a review timeline
- Outstanding issues that need to be resolved in the future and a list of ongoing studies, if any
- Documentation of any dissenting opinion(s) if consensus cannot be reached
- Operation and Maintenance requirements and how neglect alters performance
- Any ancillary benefits or unintended consequences beyond impacts on nitrogen, phosphorus and sediment loads
- A technical appendix that describes changes that will be made to the modeling and reporting tools to accommodate the BMP(s)

During the Expert Panel process, the CBP further requested recommendations for land use loading rates for the newly proposed tree canopy land uses in the Phase 6 Chesapeake Bay Watershed Model (CBWM) in August 2015. However, initial relative land use loading rate recommendations from the Expert Panel were not approved by the Water Quality GIT at their September 2015 meeting. The FWG continued development of the land use loading rates through contract work with Mr. Justin Hynicka (MD DNR) and Dr. Marion Divers (University of Pittsburgh). The Expert Panel Chair and Coordinator worked with the FWG to provide input on the development of the land use loading rates, along with review by the Expert Panel. While the charge of the Expert Panel remained focused on the UTC BMP recommendations, the panel also addressed how to best align the tree canopy land use with an annual tree canopy BMP. The WQGIT approved the relative land use loading rate for tree canopy land uses at the March 14, 2016 meeting. The final report from Hynicka and Divers (2016) is included as Appendix B to this report that provides a full description of how the loading rates for these new tree canopy land uses were derived.

SECTION 2: DEFINITIONS AND PERFORMANCE MEASURE

Definitions

The definitions for forest and tree canopy land uses are provided by the CBP Partners as part of the development of the Phase 6 CBWM and were not developed by the Expert Panel.

Forest land use: For purposes of this report, the term refers to forest land uses as defined and mapped by the CBP as part of the Phase 6 CBWM. The CBP Partners are mapping all trees at 1-m resolution using some combination of leaf-on, leaf-off, and LiDAR imagery. Generally, trees are identified in the imagery by their spectral and height characteristics with a minimum canopy area of $9m^2$ and minimum height of 5m (Chesapeake Bay Image Interpretation and Mapping Standards, Chesapeake Conservancy, and pers comm J. O'Neil-Dunne, University of Vermont, 1/21/2016). Forests include contiguous patches of trees that are greater than or equal to 1-acre, corresponding to a patch of trees with a minimum internal radius of 36m, and are generally 20m – 30m away from non-road impervious surfaces (e.g., structures, driveways, and parking lots) in developed areas and approximately 10m away from non-road impervious surfaces in rural areas. At times this report may refer to the forest land use as defined in the Phase 5.3.2 CBWM, but if not explicitly specified the report is referring to Phase 6.

Tree canopy land use: The Phase 6 CBWM includes a new set of land uses that describe tree canopy with a managed understory as developed by the CBP Partners. There are two subclasses: i) tree canopy over impervious; and ii) tree canopy over turfgrass. Generally, trees are identified in the imagery by their spectral and height characteristics with a minimum canopy area of $9m^2$ and minimum height of 5m (Chesapeake Bay Image Interpretation and Mapping Standards, Chesapeake Conservancy, and pers comm J. O'Neil-Dunne, University of Vermont, 1/21/2016). Trees are included in one of these two classes if they overtop roads, driveways, or parking lots or if they are within 20m – 30m of non-road impervious surfaces in developed areas or within 10m of non-road impervious surfaces in rural areas. The two tree canopy land uses include the majority of trees located in developed areas and all trees adjacent to rural structures. The understory is assumed to be either pervious (i.e., turfgrass) or impervious cover. Because there are no Tree Canopy land uses in the Phase 5.3.2 CBWM, this term is only used in reference to Phase 6. A very small percentage of trees are excluded from both the forest and tree canopy definitions because they are in small isolated patches, less than 1-acre after accounting for adjacent water bodies and wetlands. These trees are included in the "mixed open" land use class and are not discussed in this report.

Urban tree canopy BMP: Includes actions and/or program elements that result in expanded tree canopy through the maintenance of existing tree canopy and/or an increase in trees in the urban landscape. At this time, the actions available for credit under this BMP include urban tree planting only. Tree conservation practices that expand existing tree canopy were not considered by the Panel under this BMP, but may be considered by a future expert panel. Tree canopy that is not available for credit as part of this BMP include forest buffers and trees that are planted as part of a structural BMP (e.g. bioretention, tree bioretention/planter). This BMP will only apply starting with Phase 6 of the CBWM, replacing the Phase 5.3.2 BMP for Tree Planting (Urban).

The Chesapeake Bay Program Forestry Workgroup's BMP Verification Guidance (2014) defines **expanded tree canopy** as the overall percent of tree cover in a geographically defined locality on developed land.

Performance Measure

Credit for the proposed Phase 6 urban tree canopy BMP is based on the acres of tree canopy that will be simulated as a land use change. Applicable developed land use classes will be converted to either Tree Canopy over Impervious or Tree Canopy over Turfgrass. More details are provided in Section 3 and Appendix F. The load reduction for a land use change BMP equals the relative, or percent, load reduction of the nitrogen, phosphorus and sediment for the underlying pervious or impervious land use for the newly defined tree canopy land uses.

The credit of this BMP is cumulative, which means that the acres reported in a previous year carry over into the next year. This BMP may be considered a 'stackable' BMP, where additional BMPs may be applied to the underlying land use. For example, urban nutrient management may be applied to the pervious area under the tree canopy. As a land use change BMP, the converted acres will be eligible to receive other urban BMPs reported through NEIEN. For more information, see Appendix F.

SECTION 3: URBAN TREE CANOPY IN THE CHESAPEAKE BAY

The Panel's recommendations are only applicable to the Phase 6 Watershed Model, which is currently being reviewed and refined until the final calibration in October 2016. To better understand the Panel's recommendations described in this report, however, it is important to understand the previous Urban Tree Planting BMP that was applied in the Phase 5.3.2 Watershed Model. This section provides this context and also provides a basic overview of how the Phase 6.0 Watershed Model is expected to simulate the new Tree Canopy land uses.

3.1 How the urban tree planting BMP is simulated in the Phase 5.3.2 Watershed Model

The Phase 5.3.2 CBWM has a small number of “urban” land use classes: impervious, pervious, construction, and extractive.

The Phase 5.3.2 definition for Urban Tree Planting is as follows (CAST documentation, June 23 2015 version):

> Urban tree planting is planting trees on urban pervious areas at a rate that would produce a forest-like condition over time. The intent of the planting is to eventually convert the urban land use to forest. If the trees are planted as part of the urban landscape, with no intention to covert the area to forest, then this would not count as urban tree planting.

In addition to this definition, the FWG defined the Urban Tree Planting BMP as:

> Planting trees in an urban or residential environment with the intent to increase and sustain the tree canopy. Planting 100 trees is equivalent to converting one acre of urban land to forest. Tree replacement may need to occur but cannot be “counted” as an additional planting.

The Phase 5.3.2 BMP definition in CAST for urban tree planting converts 1 acre of urban pervious land to 1 acre of forest for every 100 trees that are planted. The basis of this definition, as described above, is that the 100 trees will produce a forest-like condition over time. While 100 trees planted in the same one acre plot could eventually produce a forest-like condition, the definition itself led to a number of problems. The interpretation of the definition for this BMP led to difficulties in regard to implementation and credit evaluation. For example, it was difficult, if not impossible, to know that reported trees are indeed planted in a contiguous acre or if those trees are individually planted throughout a local area. The CAST definition was developed specifically to only award a reduction to projects that plant at a certain density (100 trees/acre), whereas in practice there are many planting projects that do not meet that criteria.

Section 5.1 further reviews the current Phase 5.3.2 urban tree planting BMP credit.

3.2 How the tree canopy land uses will be represented in the Phase 6 Watershed Model

The Phase 6 CBWM is still under development and review at this time with final version planned for release in early 2017. A major aspect of the changes from Phase 5.3.2 to Phase 6 is the incorporation of new land uses. These land use changes have been discussed among the various CBP and WQGIT workgroups, and are not the subject of this report, but it is important to note the addition of two particular land uses under the “Developed” category of land use classes: Tree Canopy over Turfgrass and Tree Canopy over Impervious. Like all other Developed land uses, these are split three times into Regulated (MS4), Non-Regulated, and Combined Sewer System (CSS) categories, to create a total of six land uses. To account for the distribution of Tree Canopy over Roads and Tree Canopy over other impervious non-roads (building parking lots, etc.), 90% of existing canopy was assumed to be over Roads and the remaining 10% over non-road impervious surfaces for the Beta-1 version. This distribution may change for the final version of the Phase 6 CBWM as additional analysis and calibration is completed.

The Phase 5.3.2 CBWM only accounted for urban trees through the forest land use, so only areas large enough to qualify as Forest, or areas converted to forest through a land use change BMP, were captured. This potentially left many trees not meeting the definition of forested land use unaccounted for across the watershed landscape. However, in practice, the Phase 5.3.2 urban land uses would have implicitly accounted for the benefits of urban trees through the calibration process. The Forestry Workgroup identified the need to simulate urban tree canopy in a more explicit fashion to more fully account for the benefits provided by all trees and as such, credit in the Bay TMDL. Over the course of several months the CBP partnership considered the addition of tree canopy land uses, and the approved land uses included Tree Canopy over Turfgrass and Tree Canopy over Impervious. As noted above, these are then divided into three categories for tracking, reporting and planning purposes: regulated, non-regulated, and Combined Sewer System (CSS). This results in six tree canopy land uses when the two types of tree canopy are split among those three categories. Altogether, the acres of tree canopy land use in 2013-2014 represents approximately 17% of the developed acres in the Phase 6 CBWM (Beta1, January 2016), or roughly 2% of the total acres in the Chesapeake Bay watershed (Table 2).

Table 2. Preliminary estimates of tree canopy land uses[1] acreage in the Phase 6 CBWM (Beta 1 vers.)

Land Use	CSS (ac)	MS4 (ac)	Non-Regulated (ac)	Total (ac)	% Tree Land use of Developed Land uses
Tree Canopy Over Impervious	732	50,589	102,679	154,000	3
Tree Canopy Over Turfgrass	14,051	383,829	344,748	742,628	14

[1] Distribution for tree canopy land uses over impervious developed land is currently set at 90% over roads and 10% over building and other. The acreage and tree canopy proportion will likely change as the Phase 6 model goes through further evaluation prior to finalization.

SECTION 4: REVIEW OF THE AVAILABLE SCIENCE

Over 150 publications were reviewed by the Expert Panel and Hynicka and Divers (2016) to describe and quantify the hydrologic and water quality benefits associated with urban tree canopy. Of these publications, 115 publications were reviewed by the Expert Panel to evaluate the research questions defined in the scope of this Expert Panel:

1. What is the effectiveness of urban tree canopy on reducing runoff, nutrient and sediment loads?
2. How does effectiveness vary by species, over time, with differences in planting sites (e.g., distance from impervious cover or other trees, soil conditions, geographic location) and with different maintenance strategies?

Both Hynicka and Divers (2016) and the Expert Panel found a limited number of studies directly addressing the water quality benefits of urban trees. While the processes and mechanisms for reducing runoff and pollutants by trees is well known, the amount by which trees reduce runoff, and by extension improve water quality, is highly variable .Further, the results of individual studies are not directly comparable. For example, the runoff reduction and water quality benefits for urban tree canopy are more challenging to characterize given the:

- Considerable variability by species in the performance of individual trees,
- Change in effective treatment with time due to tree growth
- Ability to define contributing drainage area
- The effect of site specific conditions and climate, and
- Ability of comparison studies is limited based on type of vegetation studied.

As such, Hynicka and Divers (2016) constructed a modeling approach that assesses changes in water yield as the primary method to derive nutrient and sediment pollutant load reduction benefits for urban tree canopy (see Appendix B for full report). The literature review provided by Hynicka and Divers (2016) provides references to parameterize their water balance model and subsequent nitrogen, phosphorus and sediment relative land use loading rates. The following literature review highlights more generally, the hydrologic and water quality benefits of urban tree canopy based on the review by the Expert Panel, supplemented by work by Hynicka and Divers (2016). The data reviewed for this synthesis was not limited to the Chesapeake Bay watershed. The literature review was also extended to include studies on trees planted as a part of an urban stormwater best management practice (e.g., bioretention) to quantify the impact of urban tree canopy on water quality. The Panel also reviewed a number of studies on the water quality and runoff reduction benefits of non-urban forests, which may be considered an upper limit to any credit assigned to urban tree planting, based on the assumption that trees and forests in urban environments do not function as well as natural forests due to factors such as compacted soils, lack of understory, availability of nutrients, light and water that impact tree health. Appendix C provides the complete literature synthesis.

As trees planted in the urban riparian zone (i.e., within 100 feet of a waterbody) are currently credited under a separate best management practice (Urban Riparian Forest Buffers), this review focused primarily on the benefits of trees in upland areas.

Urban trees provide a host of other benefits, including air quality improvement, habitat for wildlife, temperature reduction and energy savings. While the focus of the Panel charge is to highlight the water quality benefits, a summary of these additional benefits are provided in Section 4.6 to acknowledge the overall beneficial impact of trees in urban environments.

4.1 Hydrologic and Water Quality Benefits

Urban tree canopy is unlike most other urban BMPs, which have a defined drainage area and are engineered to capture and remove pollutants from stormwater runoff. Trees are living and have a biological lifespan, growing beyond 100 years, with urban trees ranging on average from 19 to 60 years depending on their location (i.e., street vs residential sites) (Roman and Scatena 2011). While the processes and factors affecting the hydrological cycle and water quality are the same for all trees, the quantitative impact from urban trees, specifically, is less well-known. Extensive field-based studies on natural forests have allowed the development of numerous physical and empirical based computer models to allow broader applicability and understanding of the hydrologic and water quality impact of tree canopy.

The primary impact on water quality is attributed to the prevention of water pollution by reducing the amount of runoff generated from areas where tree canopy is present. Trees also redistribute soil-water and groundwater to the near surface from trees roots accessing water to depths greater than 2-ft (Day et al. 2010). In the absence of tree canopy, rain falling on urban surfaces such as parking lots, streets and lawns picks up various pollutants as it runs off the landscape. Therefore, the cumulative effect of tree canopy is to temporarily detain rainfall and gradually release it, regulating the flow (volume and peak) of stormwater runoff downstream and thereby preventing pollutants in rainfall and on urban surfaces from being transported to local waterways. While trees process nutrients, there are only a few permanent or long-term storage pathways as nutrients and sediment from soil, atmosphere and groundwater interact with the tree. For example, long term storage of nutrients in woody biomass, along with permanent removal of nitrogen via denitrification given favorable soil moisture, carbon and nitrogen conditions (e.g., Groffman et al. 2004, Raciti et al. 2011). The formation of insoluble compounds of phosphorus is the primary mechanism for phosphorus removal through associations with metals in the soil (Busman et al. 2002 as cited in Hynicka and Divers 2016).

The specific processes by which urban trees impact runoff are shown in Figure 1 in blue. Additional mechanisms by which trees positively influence water quality are shown in green in Figure 1, while potential contributors to runoff pollution are shown in red. When it rains, trees capture rainfall in their canopies (**rainfall interception**). Intercepted rainwater is temporarily stored in the canopy before being released by **evaporation** directly into the atmosphere or transmitted to the ground via stems, branches, and the tree trunk (**stemflow**) for root absorption. The water delivered to the base of trees penetrates the soil rapidly (**infiltration**) by following interconnected pathways in the soil formed by large roots and macropores. Rainfall that is not intercepted by the canopy later reaches the underlying ground as **throughfall.** This water can be lost to evaporation, transpiration by the underlying vegetation, or infiltration or it can become **runoff.** If the underlying ground cover is pervious, leaf litter and other organic matter, soil macropores, and small depressions all work to slow runoff, hold water and further promote infiltration. The infiltrated water can feed into local waterways through **interflow** or replenish groundwater supplies (**recharge**). In between storms, trees can also absorb water from the soil by root uptake and releases the unused portion back into the atmosphere in the form of water vapor through **transpiration**. This increases soil water storage potential, effectively lengthening the amount of time before rainfall becomes runoff.

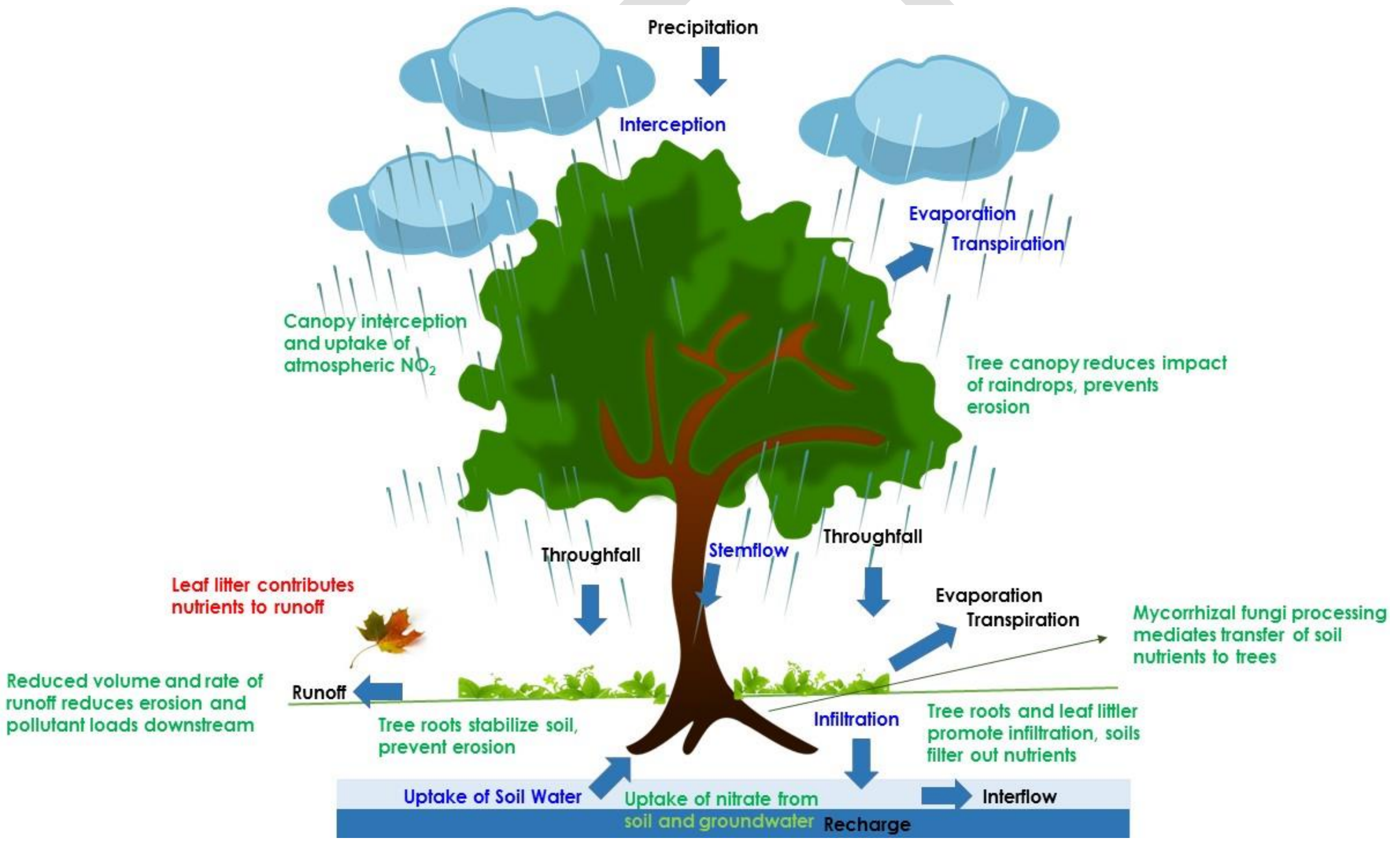


Figure 1. Urban Tree Impacts on Hydrology and Water Quality

Interception

Canopy interception of rainfall is a significant component of the tree water balance but is highly variable as it is influenced by numerous factors. For example, interception losses and storage depend on factors such as leaf area index (LAI) and tree structure, and are largely influenced by storm characteristics (Xiao et al. 2000, Keim et al. 2006). For example, the most critical time for trees to play a role in reducing runoff is during and right after a storm (KDGT 2013). KDGT (2013) suggests that, because of this, continuous simulation modeling may be the best approach for estimating rainfall interception on an annual basis. Studies of interception by urban trees report that trees can intercept 6.5-66.5% of annual rainfall, compared to 10-46% of annual rainfall for natural forests (Table 3). A more narrow range of interception values for modeling studies in the Chesapeake Bay watershed suggest annual interception values between 14.5 to 19.6% for four developed watersheds in the Piedmont, with 17% as the average (Wang et al. 2008, Band et al. 2010) (shown in bold).

Table 3. Rainfall Interception Studies of Urban Trees

Study	Location	Interception (% of annual rainfall)[1]	Species/Condition[2]	Type of Study[3]
Kirnbauer et al. 2013	Hamilton, Ontario, CA	6.5-11 17-27	G. biloba (D), P. acerifolia (D), A. saccharinum (D) L. styraciflua (D)	Modeling
Livesley et al. 2014	Melbourne, Victoria, Aus.	29 44	E. saligna (E) E. nicholii (E)	Measured
Xiao and McPherson 2002	Santa Monica, CA	27.3 15.3 66.5	All park and street trees Small jacaranda mimosifolia (D) Mature tristania conferta (E)	Modeling
Xiao et al. 1998	Sacramento County, CA	11.1	Tree canopy in the County	Modeling
Xiao et al. 2000	Davis, CA	15 27	Pear (D) Oak (E)	Measured
Xiao and McPherson 2011a	Oakland, CA	14.3 25.2 27.0	Sweetgum (D) Gingko (D) Lemon (E)	Measured
Wang et al. 2008	**Baltimore, MD**	**18.4**	**Tree canopy in Dead Run subwatershed (D)**	**Modeling**
Band et al. 2010	**Fairfax, VA**	**14.5**	**Tree canopy in Accotink watershed (D)**	**Modeling**

Study	Location	Interception (% of annual rainfall)[1]	Species/Condition[2]	Type of Study[3]
Band et al. 2010	**Baltimore, MD**	**15.7**	**Tree canopy in Gwynns Falls watershed (D)**	**Modeling**
Band et al. 2010	**Montgomery County, MD**	**19.6**	**Tree canopy in Rock Creek watershed (D)**	**Modeling**
Asadian and Weiler (2009)	Vancouver, BC	49 61	Douglas fir (E) Western red cedar (E)	Measured
Berland and Hopton 2014	Cincinnati, OH	6.7	Average value	Modeling
McPherson and Simpson 2002	Modesto, CA	3.2	Average value	Modeling
McPherson and Simpson 2002	Santa Monica, CA	7.0	Average value	Modeling
McPherson et al. 2011	Los Angeles, CA	0.4 (low) 5.6 (high)	Crapemyrtle Jacaranda (D)	Modeling
Soares et al. 2011	Lisbon, Portugal	4.5	Average value	Modeling
CWP, 2014	Montgomery County, MD	7.57	15-20 year old 9-15" DBH tree	Modeling

[1] represents the % of rain falling on the tree canopy that is captured through interception
[2] D = deciduous, E = evergreen
[3] Measured = studies that infer interception by subtracting measured throughflow and stemflow from measured rainfall; modeled = studies that model interception using models such as i-tree

Evapotranspiration (ET)

Evaporation and transpiration occur simultaneously and there is no easy way of distinguishing between the two processes. There were limited studies that quantified annual ET rates for trees in urban areas (Litvak et al. 2014, Pataki et al. 2011) and the Chesapeake Bay watershed with the exception of a modeling study by Band et al. (2010). Most studies instead evaluate how one or more factors influence ET, develop and test models for estimating ET, or measure ET values for a particular species during the growing season. Hynicka and Divers (2016) report a range of 15-25 inches/year of ET that varies based on tree age, species, canopy health and season (Ford et al. 2001; Penman 1948; Wullschleger et al. 2001; Wullschleger et al. 2000; Wilson et al. 2001; Peters et al. 2010).

In a suburban watershed in Baltimore County, MD, Band et al. (2010) identified the importance of ET on runoff reduction and noted that the major effect of tree canopy on runoff production was the ability to remove soil water by transpiration, allowing more pore space for infiltration. However, Litvak et al. (2014) found that in summer, total plot ET of urban lawns with trees was lower than lawns without trees by 0.9 – 3.9 mm d^{-1} in the Los Angeles metropolitan area. Another study from Los Angeles by Pataki et al. (2011) raised concerns that certain tree species may place too much of a demand on the local water supply because of high ET rates.

Infiltration

Studies on the effects of urban trees on soil infiltration demonstrate that trees can increase soil infiltration rates and soil hydraulic conductivity, even in highly compacted soils such as those typically found in the urban environment. Day et al. (2010) found improved structure of compacted soil by tree roots. Tree roots can increase soil infiltration rates over unplanted controls by 63% and 153% for severely compacted soils, while removal of urban forest understory and leaf litter decreased infiltration rates by 35% (Bartens et al. 2008; Kays 1980). Research findings of a biofilter by Le Coustumer et al. (2012) also suggest the importance of thick roots that help to maintain permeability of the soil over time through the creation of macropores. In non-urban environments, soil infiltration rates under tree canopy were 50% higher than outside the canopy (Mlambo et al. 2005), while deforestation or forest burning decreased infiltration rates by 20-35% (Wondzell and King 2003; Lal 1996).

Runoff Reduction

The combined effect of trees' ability to intercept and evapotranspire rainfall and promote infiltration of water into the soil is that the rate and overall proportion of rainfall that becomes runoff is reduced. Modeling studies show that, as forest cover in a municipality or watershed increases, runoff decreases (and the inverse is also true). Watershed-scale studies of runoff reduction often provide results in terms of the percent of annual runoff reduced by a given percent of tree cover in the watershed (in comparison to the runoff generated if trees were not present). These results can be translated into a percent runoff reduction per unit area of canopy if watershed areas are provided in the studies. However, not all studies are conducted on an annual basis and the results (streamflow measured at the watershed outlet) reflect not just the effect of trees in the watershed but the cumulative effect of all other land cover types and watershed features. As indicated in Table 4, the runoff reduction attributed to urban trees ranges from 2.6 to 88.8% and each study has a unique approach to quantifying runoff reduction – from individual plots or sites to watersheds. Runoff reduction attributed to natural forests is similarly wide ranging, with values from 8 to 80% from studies of water yield before and after deforestation of forested catchments.

Table 4. Studies of Runoff Reduction by Percent Tree Canopy at the Watershed or Site-Scale.

Study	Results	Description
American Forests (1999)	19% increase in runoff	Modeled increase in runoff associated with loss of 14% forest cover
Armson et al. (2013)	58% reduction in runoff in summer and 62% in winter	Measured reduction from plot containing a tree pit and surrounded by asphalt

Study	Results	Description
Wang et al. (2008)	2.6% runoff reduction	Modeled reduction associated with increasing tree cover over turf from 12 to 40%
	3.4% runoff reduction	Modeled reduction associated with increasing tree cover over impervious surface from 5 to 40%
Xiao and McPherson (2011b)	88.8% runoff reduction	Measured runoff reduction for bioswale integrating structural soils and trees[1]
Page et al. (2014)	80% runoff reduction	Measured runoff volume captured and treated by Silva Cell with tree[1]
Sanders (1986)	7% increase in runoff	Modeled increase in runoff associated with loss of 22% forest cover
	5% reduction in runoff	Modeled reduction associated with increasing tree cover over non-surfaced areas from 37% to 50%

[1] This study did not include unplanted controls

In addition to reducing total runoff volume, tree canopy can delay peak runoff because of its ability to intercept and slowly release rainfall (Asadian and Weiler 2009). Research on the ability of tree canopy to delay throughfall reports a delay in throughfall of 0.17 hours to 3.7 hours after rainfall (Asadian 2010, Xiao 2000).

Water Quality

Only one study directly addresses the effects of urban trees on the quality of stormwater runoff (shown in bold). Nine of the studies reviewed were field studies of the pollutant removal performance of stormwater treatment systems that include trees (e.g., Silva cells). However, only four of these studies (Denman 2006, Denman et al. 2011, Denman et. al 2015, Read et al. 2008) included unplanted controls to separate out the benefits provided by the tree vs. the filter media, and only one of those (Denman 2006) reported results that represent the water quality performance associated with the trees. Read et al. (2008) did not report results for trees versus other types of vegetation. Table 5 summarizes studies of pollutant removal from stormwater runoff by stormwater treatment systems with trees.

Table 5. Pollutant Removal by Stormwater Treatment Systems with Trees.

Study	Treatment System Type	Parameter and % Reduction[3]						
		TN	NOx	DIN	TKN	TP	FRP[4]	TSS
Denman 2006	**Street Tree Bioretention**	**82-95**						
Denman et. al 2011;	Biofiltration		2-78				70-96	

Denman et al. 2015								
Geronimo et al. 2014	Tree Box Filter							80-98
Page et al. 2014	Silva Cell				71, 84	72		86
Roseen et al. 2009	Street Tree			62		-54[5]		88
UNHSC, 2012	Tree Box Filter (Non-proprietary)	10		8				88
UNHSC, 2012	Filterra	15				52		85
Xiao and McPherson 2011a	Bioswale	95.3[1]						95.5[2]

[1]average of all nutrient species results
[2]average of results from TSS and TDS
[3] values represent pollutant removal associated with the entire treatment system
[4] Filterable Reactive Phosphorus
[5] Authors speculate the contribution from compost affected the results (pers.comm)

Of the other studies on water quality benefits of urban trees, a modeling study by Band et al. (2010) estimated that current tree cover in Baltimore County, MD's Baisman Run watershed reduced TSS by 981 lbs (445kg) over the simulation period, TP by 4.4 lbs (2kg), TKN by 26.5 lbs (12kg) and NO2+NO3 by 8.8 lbs (4 kg). These results were based on modeling using UFORE-Hydro that simulated changes in flow due to changes in watershed land cover, and applied national median EMC values to estimate associated changes in pollutant loads; yet, it is difficult to put these values into context because the total pollutant inputs to the watershed are unknown. Groffman et al. (2009) and (2004) illustrates the retentive nature of urban forests compared to grasslands and agricultural lands. For example, Groffman et al. (2009) found that annual nitrate leaching was higher in grass than in forest plots, except for one highly disturbed site that had hydrologic N losses well in excess of atmospheric inputs. Further, data showed an estimated N retention of 95% in a forested basin, compared to 75% for a suburban basin and 77% for an agricultural basin.

Numerous studies have evaluated the water quality benefits of natural forests. Table 6 summarizes measured nutrient and sediment exports from undisturbed forests. It also presents ratios of pollutant loading from forests that have undergone disturbance (e.g., ice damage, insect defoliation, fire) and forests that were harvested (using a range of methods such as cattle grazing, clearcutting, strip cutting, and whole tree removal) compared to the pre-disturbance or control sites for those particular studies. Given the limited amount of data on the water quality benefits of urban trees and forests, the data from undisturbed forests could be applied to establish upper bounds of pollutant removal. The ratios for disturbed and harvested forest could potentially be useful if culled to look only at studies that represent conditions commonly found in urban forest patches or planting sites (e.g., sparse cover, die-off from lack of watering, compacted soils).

Table 6. Nutrient and Sediment Loads from Non-Urban Forests[1].

Type of Forest	**Pollutant Export (lbs/acre/year)**[1] ; number of studies shown in parenthesis		
	TN	TP	TSS
Forest Land Use (CBWM 5.3.2, edge of stream)[5]	3.92	0.11	78
Undisturbed	2.14[3] (123)	0.16[2] (14)	41.9[2] (17)
	Ratio of Pollutant Export from Harvested/Disturbed Forest:Reference[4]		
Disturbed	3.09	2.04	2.04
Harvested	7.03	3.12	3.05

[1] based on studies of eastern forests compiled by Justin Hynicka from Maryland DNR for urban tree canopy land use recommendations

[2] median value

[3] calculated as the sum of median values for NO3 and TKN

[4] mean ratio of harvested or disturbed pollutant export to pollutant export from reference sites

[5] average edge of stream target

Leaf Litter

An emerging topic in urban stormwater management is the effect of nutrients and carbon from leaf litter on urban streams. Leaf litter represents a major energy source (DOC) and source of nutrients to streams where water soluble compounds readily leach from the leaves within hours to days following immersion, with macro-invertebrates and bacteria decomposing the leaf material in-stream. In urban-suburban areas, leaf litter collects in curbs and gutters and is flushed through the storm drain system, potentially contributing nutrients to urban streams.

While many urban areas have less than 40% tree canopy, the storm drainage systems provide a steady supply of leaf litter to streams, in addition to leaf fall from riparian areas (Stack et al. 2013). In a Scientific and Technical Advisory Committee Workshop report, Sample et al. (2015) report data for Baltimore, MD provided by Nowak (2014) to estimate an urban tree canopy biomass nutrient load of 28.8 lbs/ac/yr and 2.95 lbs/ac/yr of N and P, respectively for the entire canopy. In an outfall netting study in Easton, MD, Stack et al. (2013) found an average of 4.7 TN lb/ac/yr and 0.36 TP lb/ac/yr associated with leaf litter (organic gross solids) in catchments with 24% canopy cover. The difference between these loading rates is attributed in part to the aged leaf litter at the outfall and leaf litter reaching the streams compared to the total canopy used to estimate the biomass by Nowak (2014). Further, research by Hobbie et al. (2013) found rapid decomposition rates for leaves on pavement. While Hobbie et al. (2013) documented an initial loss of the leaf litter mass by 80% within 1-year, there was continual fluctuation (increase and decrease) in phosphorus.

Street sweeping studies have also quantified the potential impact of leaf litter on urban nutrient loadings. Baker et al. (2014) and Berretta et al. (2011) found that organic matter (i.e., leaf litter) comprised 10% of the mass load collected by street sweepers. Baker et al. (2014) and Waschbusch (2003) found that the leaves, or coarse organic material found or collected on the street using field sampling methods or street sweeper, contributed, 30% of the collected total phosphorus load. This ‘gutter subsidy’ was estimated by Baker et al. (2014) to be 2 lbs - 6 lbs P/curb-mile in residential catchments with up to 20% tree canopy. Further, Templer et al. (2015) estimated that up to 52 ± 17% of residential litterfall carbon (C) and nitrogen is exported through yard waste removed from the City of Boston, which is equivalent to more than half of annual N outputs as gas loss (i.e. denitrification) or leaching. The authors questioned the ramifications of the impact of such a substantial export on ecosystem services. While recent studies illustrate the available supply of leaf litter in urban areas, further research is needed to better quantify the fate, transport, and processing of leaf litter in urban watersheds and how to best account for this source as part of an urban nutrient mass balance. For example, significance of tree canopy reducing leachate of nitrogen compared to turfgrass (Nidzgorski and Hobbie 2015) and the predominance of groundwater delivery of inorganic N to streams in urban watersheds (Janke et al. 2014). Therefore, given the state of the science, Hynicka and Divers (2016) assumed a net flux of zero of nutrient from leaf litter based on a simplifying assumption of continuous cycling and distribution of nutrients as trees take up nutrients to support leaf development, through secession and litterfall.

4.2 Additional Benefits of Trees in the Urban Landscape

Trees in the urban environment provide a host of additional environmental benefits beyond the hydrologic and water quality benefits. Most notably, research demonstrates the impact of tree canopy to improve air quality, reduce temperatures in the urban environment and mitigate the effects of climate change. Small patches and linear corridors of vegetation in urban areas provide habitat and food resources supporting diverse wildlife.

Trees and their leaf area and structure are effective traps for capturing atmospheric particulates and their dispersal or movement as well. The typical large surface area to volume ratio of the tree canopy provides an extensive area for atmospheric deposition of particulates and associated pollutants. For example, Nowak et al. (2013) demonstrated that urban trees remove large amounts of fine particulate matter ($PM_{2.5}$). This modeling based study estimated that 5.2 to 72.2 tons of $PM_{2.5}$ were removed from the atmosphere in Syracuse and Atlanta, respectively, consequently improving air quality. Janhäll (2015) describes the variable effects of vegetation to disperse or concentrate air flows and pollutants in urban 'canyons", while citing that larger and denser trees greatly reduced the dispersion enhancing its function to act as sink for pollutants. In addition to particulates, trees remove gaseous air pollutants by the leaf stomata as well as surface of the trees (Nowak et al. 2006).

Trees mitigate the urban heat island (UHI) effect as their canopy shades the surface of roads, parking lots and sidewalks as well as sides of buildings. Loughner et al. (2012) for example demonstrated that adding vegetation in metropolitan areas such as Washington, D.C. and Baltimore, MD decrease surface air temperature as a result of tree shading and evapotranspiration. Even relatively sparse parking lot canopies can exert a significant cooling effect on parking lot climate and vehicle temperatures (Scott et al. 1998). Akbari et al. (1997) documented 20–45°F (11–25°C) lower temperatures for tree covered compared to unshaded areas, while trees may reduce peak summer temperatures by 2–9°F (1–5°C) (e.g., Kurn et al. 1994). Also, common to all vegetation, the adsorption of carbon dioxide affects the concentration of carbon dioxide, a notable greenhouse gas.

Trees in urban areas are an important wildlife habitat despite development patterns that leave behind or produce small areas and linear corridors of vegetation. In a study in Central Ohio, Matthew et al. (2010) find that even small patches of urban forests are often key for migrating birds, such as the Swainson's thrush, a species that is declining throughout much of its range. Abundance and diversity of bird species is also present in the most urban areas in the United States. As part of a long-term ecological study in Baltimore, MD, Nilon et al. (2011) found at least a third of the total number of bird species known to occur in the region, in the City itself. They expect their survey to under-estimate the number of species as their study did not include urban forested or park areas. The bird species included black-crowned night herons, indigo buntings, scarlet tanagers, white breasted nuthatches and a variety of warblers.

SECTION 5: PROTOCOLS TO DEFINE NUTRIENT AND SEDIMENT REMOVAL RATES

The current Phase 5.3.2 tree planting BMP definition in CAST converts 1-acre of urban pervious land for every 100 planted trees. This results in a land use conversion from urban pervious to forest land use. In practice, reported trees were converted to acres of the BMP in Phase 5.3.2 using the 100 trees/acre conversion rate. While this approach was reasonable for the Phase 5 modeling tools, significant improvements to the modeling tools under the Midpoint Assessment and in development of the Phase 6 CBWM demand an approach that gives a more realistic and scientifically-based credit reduction to tree planting efforts. Further, the resultant credit of planting a single tree or groups of trees on developed land uses would not result in a forest-like condition, thus water quality benefit, and therefore the value of credit required revision. For these reasons, among others, the Panel determined that protocols to define nutrient and sediment removal rates for the urban tree canopy BMP in Phase 6 need to consider a credit approach and value that better represented the diversity in pattern and distribution of trees (e.g. size, location and density of coverage, as well as underlying soil conditions), mortality, and how trees vary across the urban landscape – from a single tree to a small patch of trees, along a public right-of-way or in a parking lot, or in a residential yard.

In the long-term, tree planting as a type of urban tree canopy (UTC) BMP is critical to offset the loss of trees from development throughout the Bay watershed. Collectively, within the Chesapeake Bay, individuals, government and private interests need to take immediate action to expand UTC through the maintenance and protection of existing tree canopy, including forested lands, in addition to planting. Conservation actions of existing tree canopy are needed as rates of land development in the Bay are such that loss of tree canopy, largely forested lands, is outpacing the new UTC created by trees planted. Nowak (2012) estimated that 1 in every 3 trees in urban areas are planted, while two-thirds of the existing urban forest results from natural regeneration. However, natural regeneration was found to account for a much greater proportion of the influx of trees in Baltimore, MD compared to other urban areas evaluated. This new tree canopy will therefore take decades of growth to replace mature forest cover. The reality is that in the Bay watershed alone, some 750,000 acres—equivalent to 20 Districts of

Columbia—have been developed since the 1980s (Sprague et al , 2006). Over roughly the same time period, the watershed has experienced a net loss of forest land at the rate of 100 acres each day (USFS report 2006, see Figure 1). Specific to the Baltimore area, Nowak and Greenfield (2012) find that forest loss is 0.48% per year, while gains in impervious surface are 0.51% annually. Therefore, efforts to expand tree canopy in the future will depend on continued actions to plant new trees to replace lost canopy and conserve and maintain existing tree canopy that would facilitate natural generation. The protocols described below recommend an approach that recognizes these two critical factors to expand urban tree canopy and provide credit for jurisdictions towards the Bay TMDL.

The Phase 6 CBWM will include a tree canopy land use that includes: 1) tree canopy over impervious and 2) tree canopy over pervious land uses on developed (urban) lands. Current estimates of tree canopy land uses are provided in Table 2, although final estimates of this land use acreage are still under review by the Partnership until later in 2016. The current estimates represent approximately 2% of the total Bay Watershed. For land use mapping purposes, a tree is classified by the Chesapeake Bay Program methods if it has   a canopy area of 97 ft^2 ($9m^2$) and is 5 m (16.4 ft) in height (Chesapeake Bay Image Interpretation and Mapping Standards, Chesapeake Conservancy, and pers. comm. J. O'Neil-Dunne, University of Vermont, 1/21/2016).. A decision-rule is therefore recommended by the Panel to ensure the tracking and reporting of an annual expanded tree canopy BMP credit does not result in double counting tree canopy classified as a land use (see Section 6).

5.1 The Metric for Reporting the Urban Tree Canopy BMP and the Derivation of the Credit

Review of Existing Credit

The Expert Panel recommends replacing the current urban tree planting BMP used in Phase 5.3.2 of the CBWM for the Phase 6 CBWM. The existing urban tree planting BMP equates 1-acre for forested land use for every 100trees planted. The nitrogen, phosphorus and sediment reductions based on the land use conversion varies according to the calibrated land use loading rates, but the net reduction is approximately 67% for nitrogen, 77% for phosphorus and 57% for sediment based on the median Phase 5.3.2 land use loading rates for urban pervious and forest. The existing tree planting BMP is cumulative, which means that the acres reported in a previous year carry over into the next year.

The Panel found the 100 newly-planted trees equivalent to one acre conversion was too generous. The canopy cover as a result of planting 100 trees would likely not be large enough to cover one acre and likely did not account for mortality. Typically, in practice, tree planting projects 'overplant' to account for expected mortality, especially in the first few years. The proposed Phase 6 UTC BMP credit incorporates mortality into the **creditable acreage of tree canopy** to account for the fact that a proportion of trees will die and urban growing conditions are likely not optimal for the average tree to realize its full growth potential. As such, tree canopy is gained and lost year-to-year. Consequently, the

Expert Panel found no basis for equating the per acre nutrient and sediment load from urban land with 100 trees planted to that of forest. Therefore, the Panel concluded that the water quality benefits of urban tree planting should be based upon a projected growth of individual trees planted that is more representative of urban planting and growing conditions.

The Panel recommendations account for expected mortality of newly-planted trees as a way to provide a realistic, yet conservative, estimate on projected tree growth given the numerous factors that affect tree survivability and growth potential in an urban environment. Accounting for the effect of mortality on tree canopy is consistent with crediting approaches used for other urban stormwater BMPs that 'discount' the optimal function of BMP based on reasonable real-world factors that are expected to impact performance and longevity. While a 'no tree loss' assumption may provide a strong incentive for tree planting, this would likely lead to an over-estimate of actual tree canopy if the most optimistic 'gains' in tree canopy were awarded for tree planting.

The Expert Panel also considered the actual versus projected or future condition of tree canopy in development of the recommended protocol. Unlike most other stormwater BMPs, the actual water quality benefits of tree canopy are time-dependent. Trees are living and have a biological lifespan. As such, the tree canopy will increase in size as it grows each year until it peaks at maturity, then begins to decline and die. Conversely, traditional structural urban BMPs such as wet ponds will function at their engineered level immediately and if maintained will continue to function as designed for its design lifespan. For this reason, it was necessary to determine the appropriate time horizon for quantifying the benefits of an annual urban tree canopy BMP credit given the TMDL 2025 timeframe and how this additional acreage of tree canopy would affect accounting for the existing tree canopy (vs new BMP) land use acreage.

Lastly, as a result of discussions with the Chesapeake Bay Program modeling staff and Partnership, the Expert Panel assumed that the CBWM **tree canopy land use** acreage, separate from the BMP, will be updated approximately every five (5) years using high resolution remote sensing data; along with annual projections based on existing data sources (P. Claggett, USGS pers. comm.). The initial aerial mapping data is based on 2013 imagery with an expected update in 2018/19 and likely in 2023 (pers. comm., P. Claggett). Therefore, a decision rule is needed to determine what acreage of tree canopy credited as a BMP (as a result of tree planting) would be captured through these updates and what acreage of tree canopy would not have grown large enough to be classified by the high resolution mapping. The high resolution imagery has the ability to capture tree canopy 97-ft^2 in area (Chesapeake Bay Image Interpretation and Mapping Standards, Chesapeake Conservancy, and pers. comm., J. O'Neil-Dunne, University of Vermont, 1/21/2016). Appendix F describes these decision rules and integration of the BMP into the CBP modeling tools (i.e., CBWM, Scenario Builder). The end result will be that with each land use update, some acreage of tree canopy previously credited as a BMP will be assigned to one of the tree canopy land use categories if they have grown large enough to be captured by mapping, while other tree canopy will continue as an annual BMP credit.

5.2 Crediting Approach

The intent of the UTC BMP credit for tree planting is to extend the crediting of trees from forests and riparian buffers to all trees in urban areas, as they meet the proposed definition for urban tree canopy (i.e, "every tree counts"). The ability to quantify the effect of urban tree canopy on urban stormwater loads differs from other urban BMPs. Best professional judgement along with findings from research are needed to develop methods that quantify the pollutant load reductions from this BMP. While models exist to quantify the hydrological benefits of tree canopy, specifically runoff reduction, the output is highly dependent on the type of tree species, age class, climate, land use. Therefore, a method to generalize the effect of either 'classes of trees" (e.g. broadleaf, evergreen, large, medium or small trees) representative of a Chesapeake Bay default climate or sub-regions is needed for the purposes of BMP credit development.

Tree Canopy BMP Nutrient and Sediment Load Reductions

Appendix B provides the full documentation of the methods used to develop the relative land use loading rate for the tree canopy land uses. A general water balance approach (i.e., Input, I equals Output, O plus a change in storage, ΔS) is used to derive the relative effectiveness of tree canopy over turfgrass and impervious cover. This method provides an estimate of the proportion of precipitation that become surface flow (edge of field), or water yield. The water balance approach incorporates key hydrologic processes that affect the movement and fate of nutrients and sediment to include: precipitation (P), runoff (R), evapotranspiration (ET), soil leachate (L) and a change in storage term (ΔS). Hynicka and Divers (2016) describe the water mass balance in the equations below, where the subscripts *g* and *t* refer to parameters specific to turfgrass and tree canopy, respectively. Equations 4 and 5 omit the soil leachate term as this process is not applicable to impervious land cover. Rather, a laterally flowing subsurface water (T) term is introduced to quantify the water moving through the soil. This term is a necessary assumption to meet the basic physiological needs of the tree.

General Mass Balance:	$I = O + \Delta S$	**(Eq. 1)**
Turfgrass:	$P = R + ET_g + L + \Delta S$	**(Eq. 2)**
Canopy over Turfgrass:	$P = R + ET_g + ET_t + L + \Delta S$	**(Eq. 3)**
Impervious	$P + T_i = R + E + T_o + \Delta S$	**(Eq. 4)**
Canopy over Impervious:	$P + T_i = R + ET_t + T_o + \Delta S$	**(Eq. 5)**

Hynicka and Divers (2016) used daily precipitation data over a 11-year period from eight weather stations throughout the Chesapeake Bay watershed, along with representative values from the literature to parameterize the processes represented in the model (e.g. ET, L, T). The Soil Conservation Service Curve Number (CN) Method was used to estimate runoff for the four land use scenarios (i.e., turfgrass, canopy over turfgrass, impervious and canopy over turfgrass) (USDA 1989). The CN method was modified to account for the amount of precipitation adsorbed to leaves and branches after throughfall stops, effectively reducing the amount of precipitation that reaches the ground, or underlying land cover. The curve numbers used in the model were selected to represent the underlying land cover and potential modifications as a result of tree canopy overlying them.

Table 7. Curve numbers used to derive relative land use loadings rates for tree canopy (from Hynicka and Divers 2016)

Land Cover Land Use	Curve Number Value	Description
Impervious	98	Recommendation in USA Technical Release 55, Urban Hydrology for Small Watersheds
Tree Canopy over Impervious	98	Tree canopy does not modify underlying impervious cover properties
Turfgrass	79	Equivalent to turfgrass in fair conditions, HSG C
Tree Canopy over Turfgrass	74	Account for effects of canopy interception and improve physical structure of compacted soils by tree roots (see Day et al. 2010). Equivalent to turfgrass in good conditions, HSG C

This analysis provides a relative (%) load reduction for tree canopy over pervious of 23.8% for TN and TP and 5.8% for TSS. The relative load reductions for tree canopy over impervious are 8.5%, 11% and 7% for TN, TP, and TSS, respectively. These relative load reductions are applied to the underlying land cover loading rate. For example, tree canopy over turfgrass would modify the TN load reduction of turfgrass by 23.8%, or 8.5% TN load reduction of impervious loading rate where tree canopy is over impervious cover. The percent N, P and sediment load reductions for this BMP are shown in Table 8.

Table 8. Tree canopy relative land use loading rates based on the underlying land use land cover (Source: Hynicka and Divers 2016)

Land Use	Total Nitrogen Reduction (%)	Total Phosphorus Reduction (%)	Total Sediment Reduction (%)
Canopy over Turfgrass	23.8	23.8	5.8
Canopy over Impervious	8.5	11.0	7.0

The effectiveness value would then be applied to a reported acreage of new tree canopy as a result of tree planting. The total reduction would be automatically calculated through the modeling tools, but can basically be understood by the following equation:

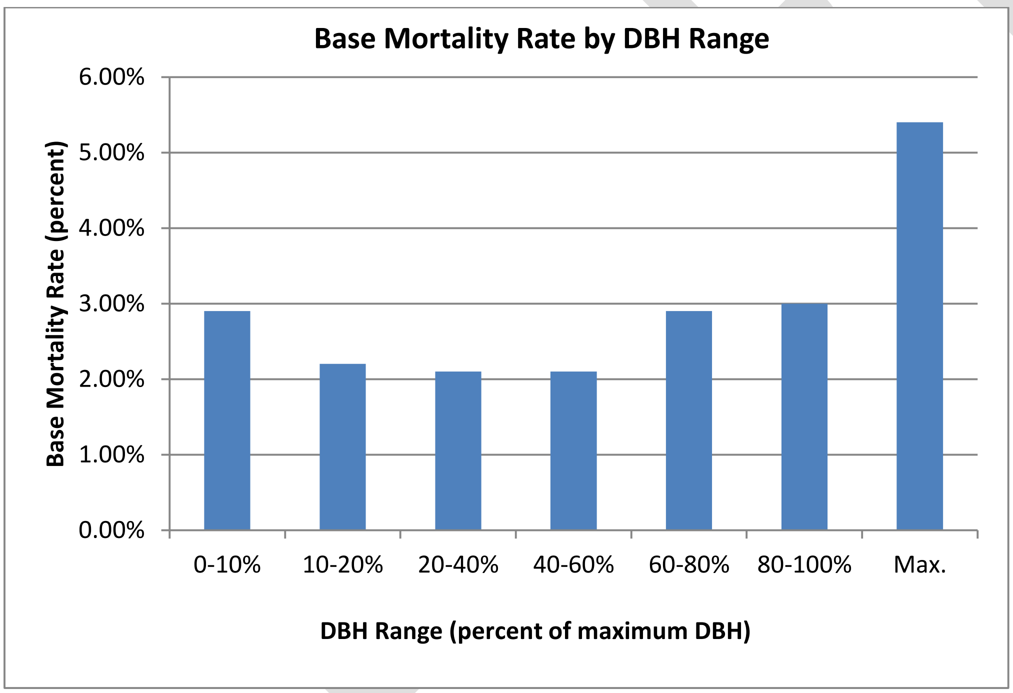
Estimated Lbs reduced/yr (edge of field)
= Tree Canopy Acreage fof Trees Planted x % loading rate reduction of TC Land Use x underlying Land Use loading rate

The central consideration in developing a method to credit tree canopy acreage for an annual BMP is the ability to translate '*number of trees planted*' to acres of tree canopy and to keep the reporting and tracking simple (e.g. # trees/acre). This desire for simplicity stems from the relatively small area treated by this BMP compared to other BMPs as reported in the Phase II WIPs progress reports, the non-structural nature of this BMP, and the numerous organizations and individuals that are involved in tree planting efforts throughout the Bay watershed.

There are numerous variables that affect the canopy size of an individual tree, and therefore the Expert Panel sought a method that could characterize these factors to provide a default or representative net tree canopy acreage based on number of trees planted. A description of the method to derive the recommended conversion factor and result is described in the following section.

i-Tree Forecast

i-Tree Forecast estimates annual tree canopy coverage amounts and growth based on tree population data for an area of interest. It is a part of the i-Tree suite of models developed by the USFS (Nowak et al. 2013a, b). Forecast is an empirical model that is planned for release in Spring 2016 as part of the i-Tree ECO model. The USFS provided simulations of Forecast based on Expert Panel input using a pre-release version of Forecast. Documentation of the model can be found in Nowak et al. (2013a, 2013b) with additional documentation expected to be available when i-Tree ECO is released (i.e., https://www.itreetools.org/resources/manuals.php). Tree cover is predicted by the following tree characteristics: species (growth rate, height at maturity), diameter at breast height (DBH), crown light exposure (CLE) and dieback. For the purposes of the simulations defined by the Panel, the growth rate is not affected by dieback as the trees planted were assumed to be in good condition. The tree characteristic data is based on data published in the literature and field data from areas throughout the United States[1]. The field and published data provide species-specific information on DBH, tree height, crown height, crown width and other variables to derive equations used in the model. The growth rate and other tree parameters of an individual tree (or group of trees) are dependent on DBH for a species, which functions as the primary independent variable in the Forecast model. Figure 2 shows how each size class of tree has a unique set of seven diameter ranges to which base mortality rates are assigned. If a user specifies a different mortality rate, a similar distribution of DBH class is used in Forecast. A user-defined mortality rate of 5% was selected by the Expert Panel (see next section for more detailed description), with additional model simulations evaluating a 2.5% mortality. This mortality rate is applied in the initial each year (at planting) but will vary in subsequent years based on DBH as shown in Figure 2 below.



[1] List of databases include: http://hort.ufl.edu/; http://plants.usda.gov; http://www.backyardgardener.com/; http://www.ces.ncsu.edu/; http://www.floridata.com; http://www.hort.uconn.edu/plants/; http://www.hortpix.com/index.html; http://en.hortipedia.com/wiki/

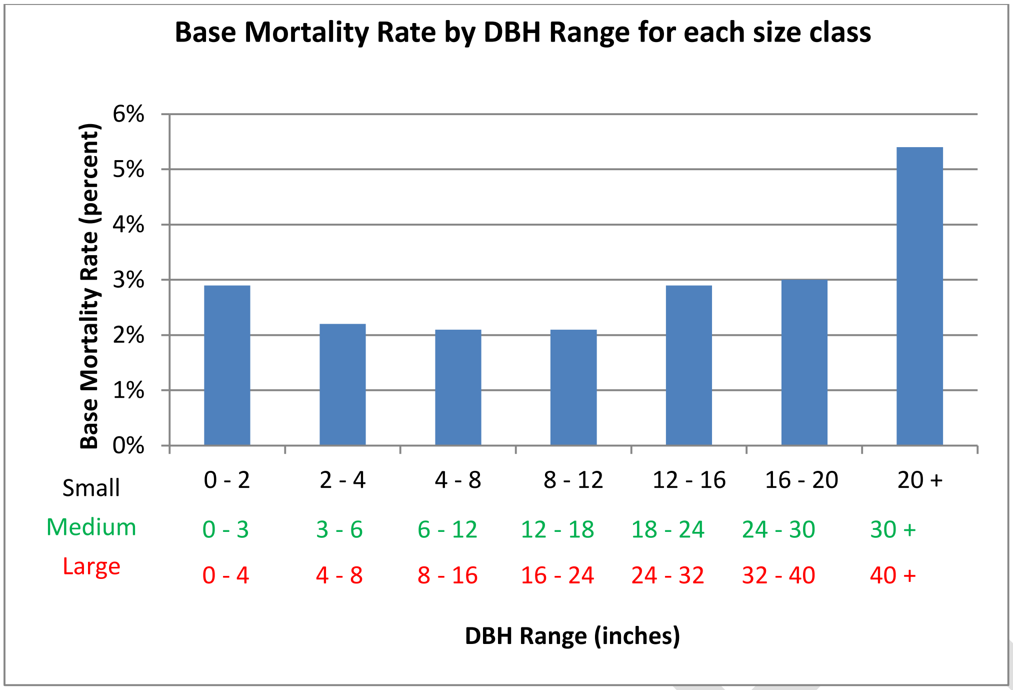


Figure 2. Mortality rate distribution by diameter class with range classified by DBH for the species (A) and for actual DBH classes for small, medium and large tree species (B), (Nowak et al. 2013b)

Description of i-Tree Forecast Simulations

Forecast simulations, for the purposes of informing Expert Panel recommendations, were designed to provide **an average annual projected canopy area for a single tree planted**. Input parameters for i-Forecast included: location (growing season length), size of tree at planting, baseline mortality rate, crown light exposure, condition of tree at planting, and tree species. The summary of the expert panel input for Forecast simulations is provided in Table 9. Twenty tree species were selected by the Panel based on commonly identified species for tree planting, to include large, medium and small broadleaf deciduous, large and small coniferous evergreen and broadleaf evergreen. The number of different species input to Forecast was balanced by a manageable number of 'cohorts' for summary output. A cohort is defined as a group of trees which all have the exact same set of input parameters. The final simulation included 40 cohorts per location (20 species, 2 crown light exposure conditions) for analysis and includes approximately 10,000 trees per cohort. The high number of trees is used as input for building precision into an average estimate of net tree canopy area for a single tree planted. The location is defined by the growing season length that is represented by the number of frost-free days (FFD) per year. Figure 3 shows the distribution of the 4 climate regions included in the simulation. The summary output also defines a default Chesapeake Bay-wide climate (166 FFD). A 5% mortality rate was initially selected by the Expert Panel based on a literature review (see Appendix C for summary of studies) and best professional judgement. Additional, scenarios used a lower mortality rate of 2.5% to evaluate the effect of average annual, projected canopy growth. In Forecast, the mortality is applied annually to the remaining tree population. For example, if 10,000 trees are planted, one year after planting 9,500 trees would remain after year 1 using a 5% mortality rate. However, the mortality rate is adjusted and lower in subsequent years as shown in Figure 2 for a given species type and DBH. In general, research finds higher mortality rates are typical of urban compared to natural forest conditions and for newly planted and younger trees. A meta-data analysis by Roman and Scatena (2011) estimated street tree annual survival rates ranged from 94.9% to 96.5% which equates to a mean life expectancy of 19-28 years. The estimated lifespan for an urban tree in Baltimore, MD was estimated to be 15 years (Nowak et al. 2004). Roman (2013) found an overall annual mortality rate of street trees in Oakland CA of 3.7%, with the highest mortality rates found for small/young trees. The same study evaluated survival of trees planted through a residential planting program in Sacramento, CA and found a survival rate of 70.9% at five years post-planting. A study of street trees in New York City found that the highest mortality rates occurred in the first few years after planting (Lu et al. 2010). The canopy area per tree planted is a metric that accounts for remaining canopy of surviving trees, divided by the original number of trees planted. The apparent decrease in canopy area per tree is because eventually the die-off/mortality rate overtakes the canopy growth rate; a modeled tree does not decrease in size, but the number of trees decreases enough that the average remaining canopy area per original tree planted does decrease.

Canopy area per tree planted is given by:

(Canopy area per surviving tree) x (Number of surviving trees) / (Number of trees originally planted)

The simulations were considered a-spatial as the only geographic reference for the simulations is climate region. Tree planting density was not addressed with the exception of light conditions specified in the simulations. For example, a park or forest-like condition would limit crown light exposure (CLE) similar to higher density planting situations. Land use was not taken into account regarding impacts on tree condition, growth rate, and mortality, as this would require location-specific input parameters at the County-level. Further, land use does not affect the projected tree canopy growth, rather environmental outputs from the model.

Table 9. Summary i-Tree Forecast model simulations.

<table>
<tr><td colspan="2">• 4 Geographic regions and one representative average (10 locations), varying number of frost-free days (FFD)/year. The ‘number for frost free days’ is used as a proxy variable to account for climate effect on growth. Other specific factors that affect tree growth explicitly such as precipitation and solar radiation are taken into account as part of modeling environmental outputs in associated hydrological and pollution models. This part of Forecast was not used by the Panel as the explicit use of Forecast was to provide an estimate for canopy growth based on trees planted.
– 105 FFD: e.g. Cooperstown, NY; Force, PA
– 150 FFD: e.g. Scranton, PA; Monterey, VA
– 166 FFD: This is the average across the Chesapeake Bay
– 210 FFD: e.g. Baltimore, MD; Norfolk, VA;
– 255 FFD: e.g. Harborton, VA (Eastern Shore, Accomack County)
• The final set of scenarios included 2 climate regions and the Chesapeake Bay default.</td></tr>
<tr><td colspan="2">• Size of tree at planting
– 1” DBH planting size was selected to be representative of tree planting projects. This size of tree is generally representative of a 5 year old tree.</td></tr>
<tr><td colspan="2">• Condition at planting
– Good condition.</td></tr>
<tr><td colspan="2">• Mortality
– Two mortality rates were simulated for analyses; 5% and 2.5% baseline mortality rate, of remaining population based on literature review values (average of 19 studies that evaluated street and/or yard trees across a wide range of geographic locations) and best professional judgement</td></tr>
<tr><td colspan="2">• Crown light exposure (2 types).
– Forecast provides two CLE parameters that differs in the amount of sunlight a crown is exposed to. A park-like setting is similar to a partially closed canopy and growth rate is reduced by 44 percent. An open-space setting received full sunlight. The simulations planted ½ of the trees in park-like setting and ½ in open space. The park-like provided reduce light exposure to a tree that would be present when trees are planted near buildings, for example.</td></tr>
<tr><td>• Trees species to represent. A total of 20 species were included in the model simulations to be representative of the different tree types:
– Broadleaf large, medium and small species (n=4, 4, 5)
– Evergreen large and medium species (n=4, 2)
– Broadleaf evergreen small specie (n=1)</td><td>Broadleaf deciduous, large
Platanus acerifolia - London planetree
Quercus michauxii - Swamp chestnut oak
Quercus phellos - Willow oak
Ulmus americana - American elm
Broadleaf deciduous, medium
Acer rubrum - Red maple
Betula nigra - River birch
Nyssa sylvatica - Black tupelo
Ostrya virginiana - Eastern hophornbeam</td></tr>
</table>

	Broadleaf deciduous, small Amelanchier arborea - Downy serviceberry Cercis canadensis - Eastern redbud Cornus kousa - Kousa dogwood Prunus yedoensis - Yoshino flowering cherry Styrax japonicus - Japanese snowbell **Coniferous evergreen, large** Cryptomeria japonica - Japanese red cedar Cupressocyparis x leylandii - Leyland cypress Pinus spp - Pine (average of genus) Pinus strobus - Eastern white pine **Coniferous evergreen, medium** Chamaecyparis thyoides -Atlantic white cedar Juniperus virginiana - Eastern red cedar **Broadleaf evergreen, small** Ilex opaca - American holly

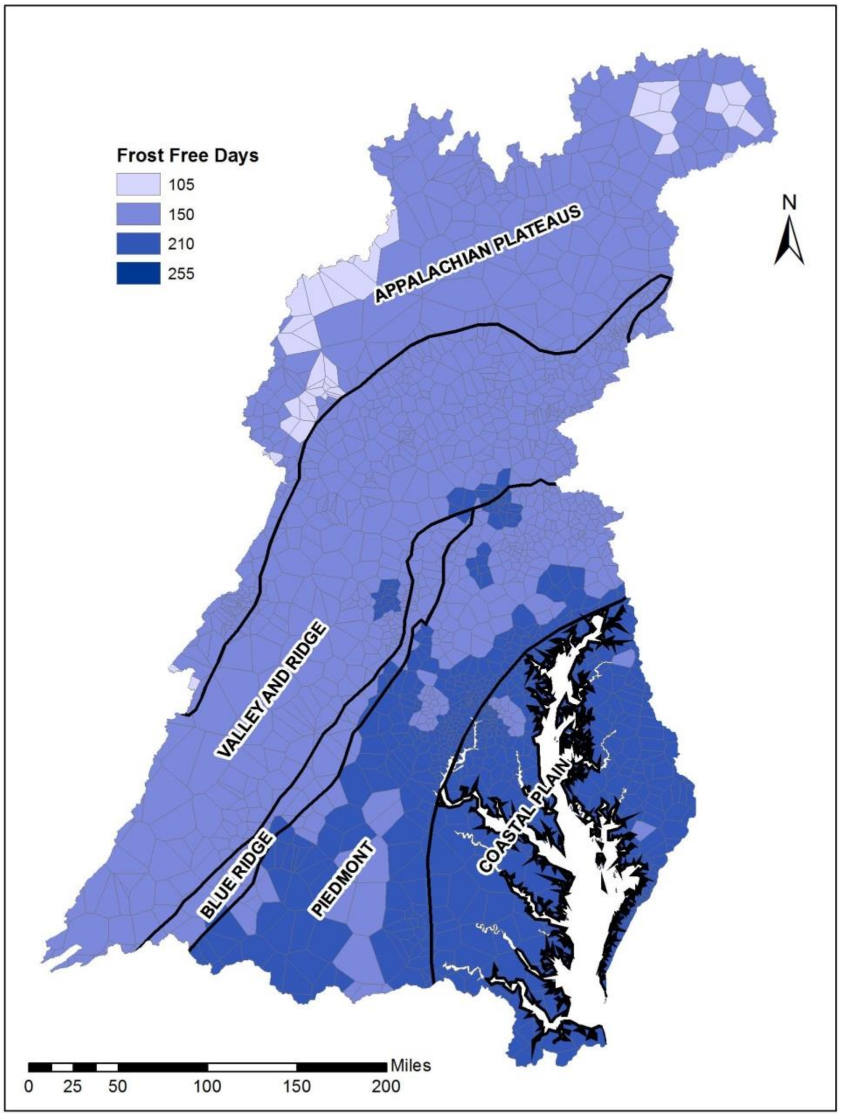


Figure 3. Climate regions used in the i-Tree Forecast model simulations

5.3 Summary of Results

Forecast provides **an average projected annual growth of tree canopy area** for a newly planted tree based on the scenarios described in Table 9. The results of the model were then used to provide an estimated acreage 'credit' for each tree planted. The results are provided for the 150 FFD, 210 FFD and Chesapeake Bay average of 166 FFD. Due to the small area of the 105 FFD and 255 FFD (3.6% of the Bay watershed area), the Panel concluded that projected canopy growth in these areas may be associated with the other 2 FFD climate areas and a Bay-wide average. The 210 FFD climate areas closely approximates the Coastal Plain and Piedmont physiographic provinces, while the 150 FFD is inclusive of the Valley-Ridge and Appalachian physiographic provinces.

Figure 4 illustrates an example set of results for all tree types simulated for the Bay-wide climate area. Additional output from Forecast is provided in Appendix E. While the growth curves for the broadleaf trees are similar, the projected canopy area for the different tree species is apparent, with evergreens and coniferous with the lowest projected canopy area and broadleaf species the greatest area. The average projected canopy area for a single tree varies by climate, where a longer growing season (i.e., more FFDs) is expected to produce a larger tree canopy. Table 9 provides results for an average, or default broadleaf tree for the three climate scenarios using a 5% and 2.5% mortality rate. The Panel provides a definition of a default or representative tree as a necessary simplification for jurisdictions to report and track this BMP given the numerous factors that affect tree canopy (i.e., growth and survivability of trees planted). The Panel's professional experience finds that coniferous evergreen trees are not commonly planted and subsequently not considered as part of the recommended credit. The default broadleaf tree is defined as projected growth for both broadleaf large and medium trees.

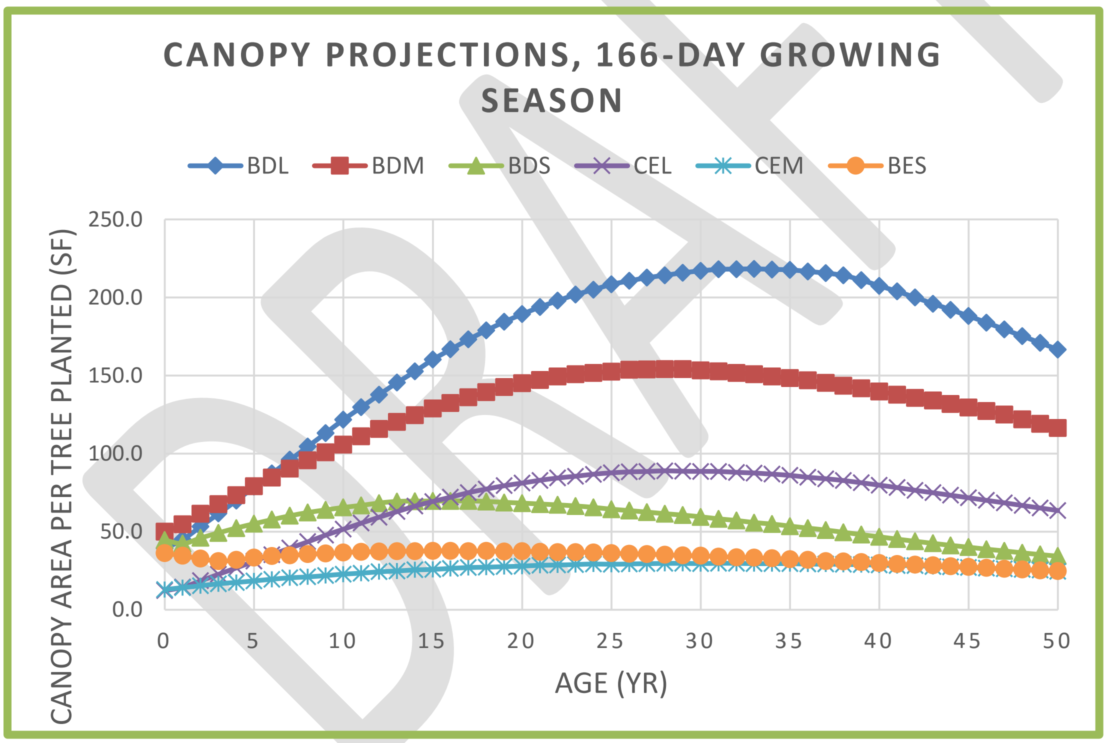


Figure 4. Canopy projections for the Chesapeake Bay-wide (166-day) growing season. (BDL = broadleaf large; BDM = broadleaf medium; BDS = broadleaf small; CEL = coniferous large; CEM = coniferous medium; BES = broadleaf evergreen)

Table 10. Projected average canopy area (ft²) for a single tree planted for specified growth period using a 5% mortality rate (a) and 2.5% mortality rate (b).

a) Default Broadleaf (ft^2), 5% mortality			
Age	**150 FFD**	**166 FFD**	**210 FFD**
	Valley Ridge and Appalachian Physiographic Region	**Chesapeake Bay-wide**	**Coastal and Piedmont Physiographic Region**
5	74	79	92
10	104	114	144
25	160	180	239

b) Default Broadleaf(ft^2), 2.5% mortality			
Age	**150 FFD**	**166 FFD**	**210 FFD**
	Valley Ridge and Appalachian Physiographic Region	**Chesapeake Bay-wide**	**Coastal and Piedmont Physiographic Region**
5	85	90	105
10	132	144	182
25	271	304	399

5.4 Final recommendation for BMP credit

The Expert Panel recognizes that new acreage of tree canopy from tree planting activities will make up a small percentage of overall tree canopy and that a net loss of tree canopy in the Chesapeake Bay watershed will continue to occur unless measures are taken to maintain and conserve existing tree canopy.

Recommendation 1: Decision Rule for Tree Canopy as a BMP and as a Land Use
In review of the Forecast results, a tree will, on average, meet the 97 sq. ft. threshold after 10 years of growth, or 10 years after planting (assuming a DBH of 1" at planting with an assumed mortality of 5%). Therefore, the recommended decision rule is that trees will require a minimum of 10 years growth after planting to reach an area necessary to be captured by high resolution imagery and mapped as a land use. The next two potential cycles for high resolution imagery mapping are 2017/18 and 2022/23. Based on this decision rule, trees planted for BMP credit in 2016 and onward will continue to be tracked as a BMP through 2025.

Recommendation 2: Lifespan of Annual BMP Credit
The lifespan of the BMP credit is based on the time period until it is mapped as a land use based on high resolution imagery (i.e., minimum of 10 years of growth after planting). This BMP would not be eligible for renewal in the National Environmental Information Exchange Network (NEIEN) once it is classified as

a land use to avoid double counting of tree canopy acreage. Unlike other structural BMPs that have a credit duration due to their expected practice lifespan, urban trees on average have an expected lifespan between 19-28 years, and longer on residential sites (Roman and Scatena 2011) .

Recommendation 3: Information for Reporting and Tracking BMP

The Panel recommends using the default broadleaf tree as a representative tree to report and track this BMP since broadleaf large and medium trees are most common for tree planting. While there are many different species planted that are typical of urban tree planting projects, the majority of the trees are either large or medium deciduous based on the Panel's experience and best professional judgement. The species selection was based on Panel input. This recommendation does not limit the type or density of trees planted that are eligible for credit. The credit applies to all tree types, whether planted individually or in a contiguous area (i.e., trees other than broadleaf species may be planted).

While climate region (FFD) is not necessary to report and track the BMP, the Expert Panel used the output from i-Tree Forecast for the Chesapeake Bay-wide climate of 166 days as guidance to determine the tree canopy acreage for credit. An assessment of the FFD shown in Figure 3 and the county jurisdictional boundaries found that many counties, specifically in the Piedmont region, may be subject to more than one FFD area. The Expert Panel concluded that reporting a canopy area credit based on multiple climate regions would be too complex and that the BMP should be as simple as possible for reporting and tracking purposes. The Expert Panel concluded that the Bay-wide FFD of 166 days should be used as guidance to project canopy growth using a a default tree to avoid complexities of multiple FFDs within a single county and provide clarity for ease of reporting and tracking.

Recommendation 4: Metric for Translating Trees Planted to Urban Tree Canopy Acreage

The Expert Panel recommends the annual BMP acreage credit be based on 10 years of projected growth after planting. The 10 years of projected growth was chosen not only to assure that trees planted can be identified through high resolution imagery but it also represents a mid-point in the projected lifespan of the tree that best aligns with the planning timeframe for the Phase III WIPs where jurisdictions must identify the BMPs that will be "in the ground" by 2025. That is, the average life expectancy of urban street trees is approximately 19-28 yrs (Roman and Scantena 2014), while Nowak (2004) estimated the average lifespan for urban trees in Baltimore, MD to be 15 years. This means that trees planted in 2016 and after will receive the full BMP credit assuming 10 years of growth whether or not they actually grow large enough by 2025. In this sense these trees are "over-credited." On the other hand, the trees that continue growing beyond 10 years will be "under-credited." Both the projected growth and the nutrient and sediment reduction credit are considered to be conservative. While the sediment and nutrient reduction credit is over-estimated during this period, after 10 years the reduction credit is under-estimated assuming proper tree care and in the absence of actions or disease that cause its mortality. This recommendation provides assurance that the continued growth of trees post 2025 will work to maintain the nutrient and sediment caps after the TMDL is met.

The Expert Panel recommends the credit for the default broadleaf tree for the Bay-wide climate is as follows **with number of trees per acre shown in parentheses**: 144 square feet of tree canopy area for every tree planted (300 trees per acre). This is based on a similar output of i-Tree Forecast for the recommended 10-years of projected growth assuming low and high mortality rates (2.5% and 5% mortality) in both the 166 and 210 FFD areas. This is the Expert Panel's best professional judgement to provide a realistic, yet conservative canopy area to credit towards the BMP.

Qualifying Conditions

The qualifying condition for this BMP is to report the number of trees planted. This is consistent with the previous urban tree planting BMP reporting requirements and information typically available given the diversity of project implementation. Jurisdictions may also report the dominant land cover on which the tree is planted (pervious or impervious). If this information is not provided, the CBP will make assumptions based on the current distribution of land uses in the Phase 6 model.

SECTION 6: ACCOUNTABILITY

Unintended Consequences and Double-Counting

The Expert Panel does not envision any unintended consequences of adopting the recommendations for this UTC BMP credit. The recommended pollutant removal efficiencies may be refined as research is further developed to quantify the nutrient impact of tree canopy on water quality, specifically in reference to leaf litter washed off of impervious cover (roads and non-roads). The current assumption applied in the derivation of the nutrient reductions for urban tree canopy is that nutrients generated by trees are transferred from the soil, throughflow, runoff and into biomass. Theoretically, this assumption results in a no net change in load. Recommendations to advance the understanding of nutrient fate from leaf litter are presented in Section 7.

The acreage of new tree canopy acreage as a result of tree planting presents a scenario for potential double-counting if a decision rule to track and report existing tree canopy mapped as a Phase 6 land use and new tree canopy from tree planting is not provided. The Expert Panel recommends the following decision rule to avoid double-counting of tree canopy acreage.

To avoid double counting with the existing tree canopy land use, annual tracking and reporting new acres of tree canopy will be reported as a land use change BMP and tracked separately from the existing land uses. The assumption of this protocol is that the tree canopy land use as part of the Chesapeake Bay Watershed Model (CBWM) is updated approximately every 5 years using high resolution imagery. The initial imagery is from 2013 and may be updated in 2018 and 2023. The BMP credit is replaced by the credit associated with the change in land use at such time when the trees planted can be recognized through high resolution imagery.

Verification

The Chesapeake Bay Program adopted verification guidance from the Forestry Workgroup for the existing tree planting BMPs. The guidance was developed based on the BMP used in the Phase 5.3.2 CBWM that does not include a tree canopy land use. Verification is an important process to ensure the BMPs implemented are still functioning as planned in order to receive continued nutrient and sediment reductions towards the Bay TMDL. In review of the existing verification guidance, the Panel determined that the FWG's current verification guidance demands a very high level of assurance that presents challenges to local jurisdictions for reporting and tracking purposes and does not align with the current methods developed to credit tree planting. Further, tree planting is conducted by many different types of organizations and groups that are either volunteer-based or associated with municipal programs, with a range of oversight and post-planting maintenance. The Expert Panel acknowledges that it is critical for tree planting projects to implement measures that ensure the greatest potential survival of trees planted. However, it is well-documented that mortality is typically high in the first few years and on average 4-6% overall. The Expert Panel strongly emphasized that it is unreasonable to assume a replanting of all trees that die in these first few years which the Verification Guidance recommends as the planting density in many cases are overplanted to account for expected mortality and the recommended crediting method accounts for a 2.5% to 5% annual mortality.

The Panel acknowledges that specific verification procedures and protocols will be determined by a jurisdiction according to their implementation priorities and programmatic capabilities in accordance with the overall BMP Verification Principles and BMP Verification Framework adopted by the CBP partnership (CBP, 2014). As such, the Panel commendations are advisory in nature, and are not binding on any State. With that in mind, the Panel asks the Partnership to consider the following for the recommended Phase 6 UTC BMP.

- The suggested BMP credit duration is 10 years. Once new high resolution imagery is available the trees will be captured through the tree canopy land uses rather than annual BMP submissions once they reach a projected growth of 97 ft^2; an objective verification procedure . It is in the interest of jurisdictions to ensure the survival of trees planted so they contribute to an expanded tree canopy and therefore improved water quality. While planted trees account for a small percentage of overall tree canopy in the Bay watershed, it is a BMP that can be widely implemented in the watershed that has tremendous value in involving multiple stakeholders in the Chesapeake Bay restoration effort. If trees are not maintained and die, then the canopy will not be captured during the land use update and seen as 'credit' lost.
- The high resolution imagery used to quantify the area of existing tree canopy land uses may be used as part of the verification process for this BMP over the long term. The high resolution imagery provides an objective process given the classification rules used by the Bay Program. While periodic updates to the high resolution land use imagery would not verify specific tree planting projects, they would verify the total tree coverage in an aggregate area. This would

provide an automated verification of total existing tree canopy and for trees 10 years after planting, although extraneous factors such as development are likely to overwhelm the effect of tree planting.

- Verification efforts by local governments would ideally focus on ensuring that management of trees planted provides for optimal survival. A 2.5% to 5% annual mortality is currently built-into the recommendations for the UTC BMP and tree replacement

Throughout its discussions, the Panel acknowledged that jurisdictions with more rigorous planting, conservation, and verification protocols for its trees will likely see greater overall gains in tree canopy relative to the Panel's recommended BMP credit. Consistent with the existing FWG verification guidance the FWG proposed to provide education to landowners about planting and after care given tree planting efforts on private lands. Put another way, a jurisdiction that better ensures the health and longevity of both its newly planted and existing trees will see accrue acres of tree canopy and forest land uses in the long run, which will be beneficial in meeting and maintaining TMDL targets in 2025 and beyond.

SECTION 7: FUTURE RESEARCH AND MANAGEMENT NEEDS

Research

- Evaluate the effect of tree canopy, non-forested lands on water quality. This may be in the form of field research or GIS-based analysis of multiple catchments that could be combined with high resolution imagery and other available data to help isolate the effect of tree canopy changes from factors that usually confound that analysis. Long-term water quality monitoring to isolate the effect on tree canopy along with advancing model development to simulate changes in urban tree canopy on water quality
- Support research to characterize and quantify the impact of leaf litter on nutrient contributions to the urban mass balance
- Continued research on the effect of soils on tree canopy growth in urban watersheds. Recent research suggests that soil volume is a key variable to urban tree growth potential (i.e, Susan Day's), along with the effect of tree canopy on altering soil physical properties such as enhanced infiltration.
- There is a need for collection of multi-year field data that explicitly measures nutrient fluxes associated with areas of tree canopy. The data should be collected in areas representative of applicable conditions within the Chesapeake Bay Watershed. The data can be used to inform future expert panels, new versions of the modeling tools or land use loading rates.
- If new data specific to nutrient fluxes from tree canopy over turf and impervious surfaces in the Chesapeake Bay watershed is not gathered, the CBP partnership can consider whether to adjust, drop, or keep this land-use/BMP as presently recommended for future model versions.

Management

- Jurisdictions review and adopt guidance for tree planting and post planting care (i.e., minimum soil volume, mulching, planting depth, appropriate tree species for location) to provide the best possible conditions and promote growth in UTC and tree health with efforts to provide (disseminate) this guidance to individuals, groups or organizations conducting tree planting.
- Jurisdictions use tools to evaluate the net loss/gain of tree canopy beyond the Chesapeake Bay land use update. It is expected to see a continued net loss despite extensive efforts to replace tree canopy by tree planting given continued development patterns in the Bay watershed
- The UTC BMP credit be reevaluated after 2025 to account for the increase in credit post 2025 as trees mature
- Develop BMP's that address the conservation and maintenance of existing tree canopy. Conservation BMP's will reduce future pollutant loads by reducing the loss of tree canopy in future land use projections. Maintaining tree canopy has the potential to have a greater positive impact on local efforts and actions to address water quality than tree planting alone.

SECTION 8: REFERENCES

Akbari, H., D. Kurn, et al. 1997. Peak power and cooling energy savings of shade trees. *Energy and Buildings* 25:139–148.

American Forests. 1999. *Regional ecosystem analysis Chesapeake Bay Region and the Baltimore-Washington Corridor: Calculating the value of nature*. American Forests. Washington, DC.

Armson, D. P. Stringer, A.R. Ennos. 2013. The effect of street trees and amenity grass on urban surface water runoff in Manchester, UK. *Urban Forestry & Urban Greening*, 12(3): 282-286.

Asadian, Y. 2010. *Rainfall interception in an urban environment*. Master of Science Thesis, the University of British Columbia. Vancouver. BC.

Asadian, Y. and M. Wieler. 2009. A new approach in measuring rainfall intercepted by urban trees in coastal British Columbia. *Water Quality Research J. Can*. 44: 16-25.

Baker, L.A. S.E. Hobbie, P. Kalinosky, R. Bintner, and C. Buyarski. 2014. Quantifying nutrient removal by street sweeping. *Final Project Report to the Minnesota Pollution Control Agency,* Sept. 2014.

Band, L., Nowak, D., Yang, Y., Endreny, T., and J. Wang. 2010. Modeling in the Chesapeake Bay Watershed: effects of trees on stream flow in the Chesapeake Bay. Report to Forest Service for Agreement No.07-CO-11242300-145.

Bartens, J, S. D. Day, J. R. Harris, Dove, J.E., and T. M. Wynn. 2008. Can urban tree roots improve infiltration through compacted subsoils for stormwater management? *Journal of Environmental Quality*, 37: 2048-2057.

Bartens, J., Day, S. D., Harris, J. R., Wynn, T. M., and J. E. Dove. 2009. Transpiration and root development of urban trees in structural soil stormwater reservoirs. *Environmental Management*, 44: 646-657.

Berland, A. and M. E. Hopton. 2014. Comparing street tree assemblages and associated stormwater benefits among communities in metropolitan Cincinnati, Ohio, USA. *Urban Forestry and Urban Greening*, 13: 734-741.

Berretta, C., S. Raje, and J. Sansalone. 2011. Quantifying nutrient loads associated with urban particulate matter and biogenic/litter recovery through current MS4 source control and maintenance practices. *Final report to Florida Stormwater Association Educational Foundation*, University of Florida. Gainesville, FL.

Boggs, J. L., and G. Sun. 2011. Urbanization alters watershed hydrology in the piedmont of North Carolina. *Ecohydrology*, 4: 256-264.

Bolton, K., and M. Greenway. 1999. Nutrient sinks in a constructed melaleuca wetland receiving secondary treated effluent. *Water Science and Technology*, 40(3): 341-347.

Bosch, J. M. and J. D. Hewlett. 1982. A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. *Journal of Hydrology*, 55: 3-23.

Busman, L., J. Lamb, G. Randall, G. Rehm, and M. Schmitt. 2002. *The Nature of Phosphorus in Soils*. University of Minnesota. (http://www.extension.umn.edu/agriculture/nutrient-management/phosphorus/the-nature-of-phosphorus/)

Center for Watershed Protection (CWP). 2014. *Impervious cover credit for individual trees.* Memo prepared for Montgomery County Department of Environmental Protection.

Cermak, J., Hruska, J., Martinkova, M., and A. Prax. 2000. Urban tree root systems and their survival near houses analyzed using ground penetrating radar and sap flow techniques. *Plant and Soil*, 219: 103-116.

Chen, Y., Zhang, Z., Li, Z., Tang, J., Caldwell, P., and W. Zhang. 2011. Biophysical control of whole tree transpiration under an urban environment in northern China. *Journal of Hydrology*, 402: 388-400.

Chen, Y., Day, S., Wick, A., and K. McGuire. 2014. Influence of urban land development and subsequent soil rehabilitation on soil aggregates, carbon, and hydraulic conductivity. *Science of the Total Environment*, 494-495: 329-336.

Cienciala, E. ; Kučera, J. ; Lindroth, A. ; Čermák, J. ; Grelle, A. ; Halldin, S. 1997. Canopy transpiration from a boreal forest in Sweden during a dry year. *Agricultural and Forest Meteorology*, 86(3): 157-167.

Crockford, R.H., and D.P. Richardson. 2000. Partitioning of rainfall into throughfall, stemflow and interception: effect of forest type, ground cover and climate. *Hydrological Processes*, 14: 2903-2920.

Day, Susan D., P. Eric Wiseman, Sarah B. Dickinson, and J. Roger Harris. 2010. Tree Root Ecology in the Urban Environment and Implications for a Sustainable Rhizosphere. *Journal of Arboriculture* 36 (5): 193.

Deguchi, A., Hattori, S., and H. Park. 2006. The influence of seasonal changes in canopy structure on interception loss: Application of the revised Gash model. *Journal of Hydrology*, 318: 80-102.

Denman, E.C. 2006. Are street trees and their soils an effective stormwater treatment measure? The 7th National Street Tree Symposium.

Denman, E.C., May, P.B., and G. M. Moore. 2015. The potential role of urban forests in removing nutrients from stormwater. Journal of Environmental Quality. Special section on The Urban Forest and Ecosystem Services.

Denman, E.C., P.B. May, and G.M. Moore. 2011. The use of trees in urban stormwater management. This paper has been presented previously at the Urban Trees Research Conference, "Trees, people and the built environment", 13 & 14 April 2011, Birmingham, UK and the ISA Annual Conference, 25 – 27 April 2011, Sydney.

Douglas, J. E., and W. T. Swank. 1972. Streamflow modification through management of eastern forests. USDA Forest Service Research Paper SE-94. USDA Forest Service, Southeastern Forest Experimental Station, Ashville NC.

Ford, C., Hubbard, R., and J. Vose. 2011. Quantifying structural and physiological controls on variation in canopy transpiration among planted pine and hardwood species in the southern Appalachians. *Ecohydrology*, 4(2): 183-195.

Geronimo, F. K. F., M.C. Maniquiz-Redillas, J.A.S. Tobio, and L.H. Kim, L H. 2014. Treatment of suspended solids and heavy metals from urban stormwater runoff by a tree box filter. *Water Science and Technology,* 69(12): 2460-7.

Goetz, S., Wright, R., Smith, A., Zinecker, E., and E. Schaub. 2003. IKONOS imagery for resource management: Tree cover, impervious surfaces and riparian buffer analyses in the mid-Atlantic region. *Remote Sensing of Environment*, 88: 195-208.

Green, S.R. 1993. Radiation balance, transpiration and photosynthesis of an isolated tree. Agricultural and Forest Meteorology, 64(3): 201-221.

Groffman, P.M., Law, N., Belt, K., Band, L.E., and G. Fisher. Nitrogen fluxes and retention in urban watershed ecosystems. *Ecosystems*, 7: 393-403.

Groffman, P. M., C.O. Williams, R. V. Pouyat, L.E. Band, and I.D. Yesilonis. 2009. Nitrate leaching and nitrous oxide flux in urban forests and grasslands. *Journal of Environmental Quality,* 38(5): 1848-60.

Guevara-Escobar, A.; E. González-Sosa, C. Véliz-Chávez, E. Ventura-Ramos, and M. Ramos-Salinas. 2007. Rainfall interception and distribution patterns of gross precipitation around an isolated Ficus benjamina tree in an urban area. *Journal of Hydrology*, 333: 532-541

Guidi, W., Piccioni, E., and E. Bonari. 2008. Evapotranspiration and crop coefficient of poplar and willow short-rotation coppice used as a vegetation filter. *Bioresource Technology*, 99: 4832-4840.

Heal, K.V., Stidson, R.T., Dickey, C.A., Cape, J.N., and M.R. Heal. 2004. New data for water losses from mature Sitka spruce plantations in temperate upland catchments. Hydrological Sciences Journal, 49(3): 477-493.

Herrera Environmental Consultants. 2008. *The Effects of Trees on Stormwater Runoff*. Prepared for Seattle Public Utilities. Herrera Environmental Consultants, Seattle, WA.

Hibbert, A.R. 1969. Water yield changes after converting a forested catchment to grass. *Water Resources Research*, 5 (3): 634-640.

Hirschman, D. Collins, K., and T. Schueler. 2008. The runoff reduction method: technical memorandum. Center for Watershed Protection & Chesapeake Stormwater Network, Ellicott City, MD. Pages 1-25.

Hornbeck, J. W., Martin, C. W., and C. Eager. 1997. Summary of water yield experiments at Hubbard Brook Experimental Forest, New Hampshire. *Canadian Journal of Forest Research*, 27(12): 2043-2052.

Inkilainen, Elina, M. R. McHale, G. B. Blank, A. L. James, and E. Nikinmaa. 2013. The role of the residential urban forest in regulating throughfall: A case study in Raleigh, North Carolina, USA. *Landscape and Urban Planning*, 119: 91-103.

Interstate Technology Regulatory Cooperation (ITRC). 2001. Technical/Regulatory Guidelines: Phytotechnology Technical and Regulatory Guidance Document. ITRC.

Janhäll, S. 2015. Review on urban vegetation and particle air pollution – Deposition and dispersion. *Atmospheric Environment.* 105: 130-137.

Johnson, S.R., M. R Burchell III, R. O Evans, D. L.Osmond and J. W. Gilliam. 2013. Riparian buffer located in an upload landscape position does not enhance nitrate-nitrogen removal. *Ecological Engineering*, 52:252-261.

Jones, J.A. 2000. Hydrologic processes and peak discharge response to forest removal, regrowth, and roads in 10 small experimental basins, western Cascades, Oregon. *Water Resources Research.* 36(9): 2621-2642.

Kays, B.L. 1980. Relationship of forest destruction and soil disturbance to increased flooding in suburban North Carolina piedmont. Proceedings of the Third Conference of the Metropolitan tree Improvement Alliance, 118-125. The Metropolitan Tree Improvement Alliance.

Kestrel Design Group Team (KDGT). 2013. The Roles and Effects of Tree Evapotranspiration and Canopy Interception in Stormwater Management Systems and Strategies. MPCA Stormwater Manual Revisions. Prepared by the Kestrel Design Group Team for the Minnesota Pollution Control Agency.

Kirnbauer, M.C., B.W. Baetz, W.A. Kenney. 2013. Estimating the stormwater attenuation benefits derived from planting four monoculture species of deciduous trees on vacant and underutilized urban land parcels. *Urban Forestry and Urban Greening*, 12(3): 401-407.

Kjelgren, R. and T. Montague. 1998. Urban Tree Transpiration over Turf and Asphalt Surfaces. *Atmospheric Environment*. 32(1): 35-41.

Kurn, D., S. Bretz, B. Huang, and H. Akbari. 1994. The Potential for Reducing Urban Air Temperatures and Energy Consumption through Vegetative Cooling . *ACEEE Summer Study on Energy Efficiency in Buildings, American Council for an Energy Efficient Economy.* Pacific Grove, California.

Lal, R. 1996. Deforestation and land use effects on soil degradation and rehabilitation in western Nigeria I. soil physical and hydrological properties. *Land Degradation and Development*, 7: 19-45.

Le Coustumer, S., Fletcher, T. D., Deletic, A., Barruad, S., and P. Poelsma. 2012. The influence of design parameters on clogging of stormwater biofilters: a large-scale column study. *Water Research*, 46: 6743-6752.

Link, T.E., Unsworth, M., and D. Marks. 2004. The dynamics of rainfall interception by a seasonal temperate rainforest. *Agricultural and Forest Meteorology*, 124: 171-191.

Litvak, E., Bijoor, N. S., and D. E. Pataki. 2014. Adding trees to irrigated turfgrass lawns may be a water-saving measure in semi-arid environments. Ecohydrology, 7: 1314-1330.

Livesley, S.J., B. Baudinette and D. Glover. 2014. Rainfall interception and stem flow by eucalypt street trees – The impacts of canopy density and bark type. *Urban Forestry and Urban Greening,* 13(1): 192-197.

Loughner, C.P., D. J. Allen, D. Zhang, K. E. Pickering, R. R. Dickerson and L. Landry. 2012. Roles of urban tree canopy and buildings in urban heat island effects: parameterization and preliminary results. American Meteorological Society. DOI: http://dx.doi.org/10.1175/JAMC-D-11-0228.1

Matthews, S. N. and P. G. Rodewald. 2010. Movement behaviour of a forest songbird in an urbanized landscape: the relative importance of patch-level effects and body condition during migratory stopover. Landscape Ecology, 25(6):955-965

Matteo, M. T. Randhir and D. Bloniarz. 2006. Watershed-Scale Impacts of Forest Buffers on Water Quality and Runoff in Urbanizing Environment. *Journal of Water Resources Planning and Management,* 132(2): 144-15.

McPherson, G.E. and J. R. Simpson. 2002. A comparison of municipal forest benefits and costs in Modesto and Santa Monica, California, USA. *Urban Forestry & Urban Greening*, 11: 61–74.

McPherson, G., J. R. Simpson, Q. Xiao, and C. Wu. 2011. Million Trees Los Angeles canopy cover and benefit assessment. *Landscape and Urban Planning*, 99: 40-50.

Metro. 2002. Green streets: Innovative solutions for stormwater and stream crossings. Portland, OR. 144 p.

Mlambo, D., Nyathi, P., and I. Mapaure. 2005. Influence of *Colophospermum mopane* on surface soil properties and understorey vegetation in a southern African savanna. *Forest Ecology and Management*, 212: 394-404.

Molchanov, A. A. 1960. Gidrologicheskayo rol lesa (Hydrological role of forest). Izd-vo AN SSSR, Moscow, 488 pp.

Moore, R.D., and S.M. Wondzell. 2005. Physical hydrology and the effects of forest hydrology in the Pacific Northwest: a review. *Journal of the American Water Resources Association*, 41(4): 763-784.

Nilon, C.H., P.S. Warren and J. Wolf. 2011. Baltimore birdscape study: identifying habitat and land-cover variables for an urban bird-monitoring project. Urban Habitats. Vol. 6 (1).

Nowak, D. 2014. Personal communication. Urban tree canopy analysis for Baltimore City. As cited in Sample et al. 2015.

Nowak, D., S. Hirabayashi, A. Bodine and R. Hoehn. 2013. Modeled $PM_{2.5}$ removal by trees in ten U.S. cities and associated health effects. *Environmental Pollution.* 178: 395-402.

Nowak, D. et al 2013a. Assessing Urban Forest Effects and Values: the Greater Kansas City Region. US Forest Service, Newtown Square, PA.

Nowak, D. et al 2013b. Assessing Urban Forest Effects and Values: Toronto's Urban Forest. US Forest Service, Newtown Square, PA.

Nowak, D. 2012. Contrasting natural regeneration and tree planting in fourteen North American cities. *Urban Forestry & Urban Greening*. 11: 374-382.

Nowak, D., D. E. Crane and J. C. Stevens. 2006. Air pollution removal by urban trees and shrubs in the United States. *Urban Forestry & Urban Greening*. 4: 115-123.

Nowak, D.J., Kuroda, M., and Crane, M.E. 2004. Tree mortality rates and tree population projections in Baltimore, Maryland, USA. *Urban Forestry & Urban Greening*, 2: 139-147.

Page, J.L., R.J. Winston, and W.F. Hunt, III. 2014. Soils beneath pavements: An opportunity for stormwater control and treatment. *Ecological Engineering*. In press
Read, J., T. Wevill, T. Fletcher and A. Deletic. 2008. Variation among plant species in pollutant removal from stormwater in biofiltration systems. *Water Research*, 42(4-5): 893-902.

Pataki, D. E., Carreiro, M. M., Cherrier, J., Grulke, N. E., Jennings, V., Pincetl, S., Pouyat, R. V., Whitlow, T. H., and W. C. Zipperer. 2011. Coupling biogeochemical cycles in urban environments: ecosystem services, green solutions, and misconceptions*. Frontiers in Ecology and the Environment*, 9 (1): 27-36.

Peters, E. B., McFadden, J. P., and R. A. Montgomery. 2010. Biological and environmental controls on tree transpiration in a suburban landscape. *Journal of Geophysical Research*, 115(4).

Post, D.A., and J.A. Jones. 2001. Hydrologic regimes of forested, mountainous, headwater basins in New Hampshire, North Carolina, Oregon, and Puerto Rico. *Advances in Water Resources*, 24: 1195-1210.

Read, J., Wevill, T., Fletcher, T., and A. Deletic. 2008. Variation among plant species in pollutant removal from stormwater in biofiltration systems. *Water Research*, 42: 893-902.

Reynolds, E.R.C., F.B. Thompson, and United Nations University. 1988. Forests, Climate, and Hydrology: Regional Impacts. United Nations University, Tokyo, Japan. http://www.nzdl.org/fast-cgi-bin/library?e=d-00000-00---off-0unu--00-0--0-10-0---0---0prompt-10---4-------0-1l--11-en-50---20-about---00-0-1-00-0-0-11-1-0utfZz-8-00&cl=CL2.1&d=HASH4845310c3238b42b2d2574&gt=2.

Roseen, R., T.P. Ballestero, J.J. Houle, P. Avellaneda , J. Briggs, G. Fowler, and R. Wildey. 2009. Seasonal performance variations for storm-water management systems in cold climate conditions. *Journal of Environmental Engineering*. 135:128-137.

Sample, D., K. Berger, P. Claggett, J. Tribo, N. Goulet, B. Stack, S. Claggett, and T. Schueler. 2015. The peculiarities of pervious cover: A research synthesis on allocating pollutant loads to urban land uses in the Chesapeake Bay. STAC Publication Number 15-001, Edgewater, MD. 57 pp.

Sanders, R. 1986. Urban vegetation impacts on the hydrology of Dayton, Ohio. *Urban Ecology*, 9: 361-376

Schueler, T.R., 1987. Controlling urban runoff: a practical manual for planning and designing urban BMPs. Publication No. 87703. Metropolitan Washington Council of Governments, Washington, DC.

Shaw, D. and Schmidt, R. 2003. Plants for stormwater design. Species selection for the Upper Midwest. St. Paul, MN: Minnesota Pollution Control Agency. 369 p.

Sinclair, T.R., Holbrook, N.M., and N. Zwieniecki. 2005. Daily transpiration rates of woody species on drying soil. *Tree Physiology*, 25: 1469-1472.

Soares, A.L., Rego, F.C., McPherson, E.G., Simpson, J.R., Peper, P.J., and Q. Xiao. 2011. Benefits and costs of street trees in Lisbon, Portugal. *Urban Forestry & Urban Greening*, 10: 69-78.

Sprague, E., D. Burke, S. Claggett and A. Todd. 2006. *The State of the Chesapeake Forests.* Conservation Fund, Arlington, VA.

Stack, B., N. Law, S. Drescher and B. Wolinski. 2013. Gross Solids characterization study in the Tred Avon Watershed Talbot County, MD. Prepared for Talbot County Department of Public Works. Center for Watershed Protection. Ellicott City, MD.

Tedela, N.H., McCutcheon, S.C., Rasmussen, T.C., Hawkins, R.H., Swank, W.T., Campbell, J.L., Adams, M.B., Jackson, C.R., and E.W. Tollner. 2012. Runoff curve numbers for 10 small forested watersheds in the mountains of the eastern United States. *Journal of Hydrologic Engineering* 17: 1188-1198.

Templer, P. H., Toll, J. W., Hutyra, L. R., and S. M. Raciti. 2015. Nitrogen and carbon export from urban areas through removal and export of litterfall. *Environmental Pollution*, 197: 256-261.

United States Department of Agriculture Soil Conservation Service (USDA SCS). 1986. Urban hydrology for small watersheds. Technical Release 55 (TR-55).

University of New Hampshire Stormwater Center. 2012. University of New Hampshire Stormwater Center 2012 Biennial Report.

Urban, J. 1999. Room to grow. *Treelink* 11:1-4.

USDA, Natural Resources Conservation Service, Conservation Enginering Divisions. 1989. “Urban Hydrology for Small Watersheds.” *TR-55*.

Wang, J., T.A. Endreny, and D. J.Nowak. 2008. Mechanistic Simulation of Tree Effects in an Urban Water Balance Model. *Journal of the American Water Resources Association*, 44(1): 75-85.

Wang, H., Ouyang, Z., Chen, W., Wang, X., Zheng, H., and Y. Ren. 2011. Water, heat, and airborne pollutants effects on transpiration of urban trees. *Environmental Pollution*, 159: 2127-2137.

Wang, H. 2012. Transpiration rates of urban trees, *Aesculus chinensis*. *Journal of Environmental Sciences*, 24(7): 1278-1287.

Waschbush, R.J. 2003. Data and Methods of a 1999-2000 Street Sweeping Study on an Urban Freeway in Milwaukee County, Wisconsin. Open File Report 03-93. U.S. Department of the Interior, U.S. Geological Survey.

Wondzell, S., and J. King. 2003. Postfire erosional processes in the Pacific Northwest and Rocky Mountain regions. *Forest Ecology and Management*, 178: 75-87.

Wullschleger, S. D., Wilson, K. B., and P. J. Hanson. 2000. Environmental control of whole-plant transpiration, canopy conductance, and estimates of the decoupling coefficient for large red maple trees. *Agricultural and Forest Meteorology*, 104: 157-168.

Wullschleger, S. D., Hanson, P. J., and D. E. Todd. 200. Transpiration from a multi-species deciduous forest as estimated by xylem sap flow techniques. *Forest Ecology and Management*, 143: 205-213

Xiao, Q., E.G. McPherson, J.R. Simpson, and S.L. Ustin. 1998. Rainfall Interception by Sacramento's Urban Forest. *Journal of Arboriculture,* 24(4): 235-244.

Xiao, Qingfu, E.G. McPherson, S.L. Ustin, M.E. Grismer. 2000. A new approach to modeling tree rainfall interception. *Journal of Geophysical Research*, 105(D23): 29, 173-29,188.

Xiao, Q and E.G. McPherson. 2002. Rainfall Interception by Santa Monica's Municipal Urban Forest. *Urban Ecosystems*, 6: 291-302.

Xiao, Q. and E.G. McPherson. 2011a. Performance of engineered soil and trees in a parking lot bioswale. *Urban Water Journal,* 8(4): 241-253.

Xiao, Q. and E.G. McPherson. 2011b. Rainfall interception of three trees in Oakland, California. *Urban Ecosystems*, 14: 755-769.

Yang, J.L., G.L. Zhang. 2011. Water infiltration in urban soils and its effects on the quantity and quality of runoff. *J Soils Sediments*.11: 751-761.

Zhang, W., Y. Youbin; Y. Tong, L. Ou, D. Hu, and X. Wang. 2011. Contribution and loading estimation of organochlorine pesticides from rain and canopy throughfall to runoff in an urban environment. J*ournal of Hazardous Materials*, 185(2-3): 801-806.

Appendix A: Panel Meeting Minutes

Provided as a separate document for duration of CBP partnership review and comment. Will be included with full length report following WQGIT approval.

Appendix B: Relative Reductions in non-point source pollution loads by urban trees.

Relative reductions in non-point source pollution loads by urban trees

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Background

Trees modify the fate and transport of water, nutrients, and sediment in natural and developed landscapes due to their unique physical structure compared to other plant species, basic physiological processes, and long lifespan. Since 2003, it has been the policy of the Chesapeake Bay Program (CBP) partners to increase tree canopy cover for water quality and other benefits. While tree planting best management practices (BMPs) provide opportunities to account for the water quality benefits of new trees, the benefits of existing urban trees that do not meet the definition of forest are not directly accounted for in the CBP model land uses. Advances in remote mapping technology now allow tree canopy to be mapped and total cover quantified at small spatial scales (O'Neil-Dunne et al. 2014), providing the basic information needed to ascribe water quality benefits to all urban trees of a minimum canopy size.

As the spatial resolution of tree canopy mapping increases, the land use characteristics and management decisions beneath the canopy of a tree broaden. In particular, many urban trees likely lack an understory of herbaceous and woody plants, lack a distinct organic soil horizon, are surrounded by soils affected by urban development and compaction, and have an increased probability of fertilization. For these reasons, it is not appropriate to apply the loading rate of forests – the lowest non-point source pollution loading rate among CBP land uses – to urban trees. Here we describe how trees attenuate and store non-point source pollution and describe a method for estimating unique relative pollution loading rates for two new CBP land uses: tree canopy over turfgrass and tree canopy over impervious surfaces.

Attenuation and removal of non-point source pollution by trees

It is well known that nitrogen and phosphorus are essential plant nutrients, and there is strong evidence that organic matter containing nitrogen accumulates in soils beneath trees over time. On a per-hectare basis forests store more carbon annually than urban open space due to higher tree densities. Yet, on a per tree basis urban trees may fare better, the same, or slightly worse than trees in a forest due to the release of competition for resources and/or different environmental stresses (McHale et al. 2009). Later we show that average annual uptake of N and P per tree ranges from ~ 0.31 to 1.20 lbs. and 0.17 to 0.60 lbs., respectively. Trees, and in particular the root system, increase soil organic matter contents by physically incorporating organic matter, adding plant litter to carbon stores, exuding carbon from the roots, and root die-off (Day, Wiseman et al. 2010). Soil under urban trees has been shown to have a higher carbon content than bare soil or soil under grass (Takahashi et al. 2008; Huyler et al. 2014), therefore allowing greater potential for nutrient uptake and cycling (Jo and McPherson 1995). Carbon cycling, in turn, drives N retention via uptake and microbial utilization (Lovett, Weathers et al. 2002).

In addition to storing nitrogen in biomass and soils, trees promote biogeochemical processes that convert biologically active nitrogen to inactive forms. Nitrogen inputs include organic nitrogen, nitrogen oxide gases, and ammonium nitrate particles that can be deposited as both wet and dry deposition. These inputs of nitrogen can be transformed through biological processes in soil into nitrate-N, a highly soluble form that leaches from through soils and easily transported by groundwater flowpaths (Wakida and Lerner 2005). Conversely, nitrate-N can undergo denitrification in urban systems (Groffman et al. 2004), a microbial process that transforms nitrate-N into inert dinitrogen gas and returns nitrogen to the atmosphere.

Removal of nitrate-N via denitrification is strongly influenced by soil moisture (Groffman et al. 2004; Kaushal et al. 2008). Trees promote greater soil moisture by creating preferential hydrologic flow paths and absorbing water via root systems (Day, Wiseman et al. 2010). In urban areas, turfgrass roots are likely not reaching groundwater, while deeper-rooted trees may uptake and re-distribute soil moisture from farther belowground upwards toward the surface (Day, Wiseman et al. 2010). This assists the plant community by promoting growth and ultimately increasing nutrient utilization. Greater soil moisture near the land surface, where carbon and nitrogen are also concentrated, may also promote greater rates of denitrification (Gift et al. 2010; Groffman et al. 2004; Raciti et al. 2011; Zhu, Dillard, and Grimm 2005).

In contrast to nitrogen-N, the primary form of phosphorus (orthophosphate) is less soluble and primary transported from landscapes to surface water in overland flow. Phosphorus losses from landscapes have been shown to be strongly dependent on runoff, which can transport phosphorus in particulate and dissolved forms (Staver and Brinsfield 2001). Phosphorus is generally not a major component in groundwater because it is in forms that will be bound to soil particles (Correll 1998) and most P has been shown to remain in the top three feet of soil (Daniels et al. 2010).

The primary retention mechanism for phosphorus is from irreversible binding of phosphorus to soil clay minerals. Over time, orthophosphate forms insoluble compounds through associations with metals such as iron, aluminum, and calcium in the soil (Busman, Lamb et al. 2002). These fixed forms of phosphorus are generally unavailable to plants. Therefore, retention of P is best achieved by increasing infiltration of water, bringing the nutrients in that water into contact with soil, and the formation of insoluble compounds in the soil. Tree roots reduce soil compaction and create soil macropores that increase water infiltration. Increased infiltration leads to greater pollutant/soil interaction. Additionally, trees remove soil water through evapotranspiration, thereby increasing storage capacity of soil water for future precipitation (Day, Wiseman et al. 2010).

Lastly, sediment sources in urban areas include the particles from regional and local atmospheric deposition. Trees have been shown to increase trapping of particles when compared to grass, a difference attributed to reduced overland flow, allowing particles to settle out via sedimentation (Leguedois, Ellis et al. 2008). As more water is retained beneath urban trees, the majority of sediments that would otherwise be transported by that water are likely to be retained and added to soil stores via adsorption onto organic matter and soil surfaces (Leguedois, Ellis et al. 2008).

Absolute versus relative loading rate calculations

Despite strong evidence that trees promote the attenuation and removal of non-point source pollution, we know of no studies that have quantified absolute loading rates for urban tree canopy land uses. There is very limited pollution concentration data available for runoff and subsurface flow beneath urban tree canopies that can be used to calculate representative, statistical, edge of field (EOF) loading rates for nitrogen, phosphorus, and/or sediment. This is in contrast to forested watersheds where surface water quantity and quality can be more easily evaluated (for example see: Campbell et al. 2004). Furthermore, pollution fluxes determined from streams in uniform watersheds integrates many biotic and abiotic processes from upland to riparian areas (Vannote et al. 1980). Because urban tree canopy land uses are small by definition, it is unlikely that non-point source pollution in surface water can be traced back to them in complex urban watersheds especially where larger patches of forest may also exist.

Spatial and temporal variation of inputs also makes quantifying absolute loading rates challenging among all CBP land uses. Atmospheric deposition and fertilizer are the primary inputs of nitrogen, phosphorus, and sediment to developed areas. Total atmospheric deposition (i.e., wet and dry deposition) is well documented and varies regionally over the Chesapeake Bay watershed (Linker, Dennis et al. 2013), and is often elevated in densely developed areas, relative to agricultural, forested, and suburban areas (Lovett, Traynor et al. 2000). Atmospherically sourced phosphorus is generally deposited as dust or aerosol (Correll 1998), while fertilizer includes orthophosphate and polyphosphate (which readily converts to orthophosphate when in contact with water). Variation in turfgrass fertilization rates have been documented (Aveni, Berger et al. 2013), but are extremely difficult to predict spatially. Fertilization is one of the primary reasons that nutrient loads from turfgrass are elevated in the Chesapeake Bay model relative to open space (~ 5x and 2x for N and P, respectively) and forests (~7x and 12x for N and P, respectively).

No matter whether the source of nutrients is from atmospheric deposition or fertilizer, hydrologic processes ultimately govern the fate and transport of non-point source pollution to surface waters. Therefore, given (1) the lack of data needed to estimate absolute pollution loads of tree canopy land uses and (2) high spatial variability in pollution inputs, we choose to estimate reductions in non-point source pollutant loads by trees by modeling changes in water yield relative to turfgrass and impervious surfaces – the land cover types beneath tree canopy. While we have already described how trees store and aid in pollution removal, this approach nevertheless relies on our best professional judgment that changes in water quality are proportional to changes in water yield.

Methods

We used fundamental principles in watershed hydrology to estimate the relative reduction in water yield by tree canopy compared to two underlying land uses – turfgrass and impervious surfaces – over which tree canopy extends and can be mapped using remote sensing technology. A general water balance equation (Eq. 1) is shown below where I is inputs, O is outputs, and ΔS is the change in water stored in soils. For comparison, water balance equations (Eq. 2 to 5) for two existing CBP land uses and those land uses with tree canopy are also shown. In these equations, inputs include precipitation (P), and/or laterally flowing subsurface water (T_i, throughflow), and outputs include runoff (R), evapotranspiration (ET), gravitational soil water that drains beneath the plant rooting zone (i.e., leaching, L), evaporation (E), and throughflow (T_o). The subscripts g and t refer to parameters specific to turfgrass and tree canopy, respectively.

General Mass Balance:	$I = O + \Delta S$	**(Eq. 1)**
Turfgrass:	$P = R + ET_g + L + \Delta S$	**(Eq. 2)**
Canopy over Turfgrass:	$P = R + ET_g + ET_t + L + \Delta S$	**(Eq. 3)**
Impervious	$P + T_i = R + E + T_o + \Delta S$	**(Eq. 4)**
Canopy over Impervious:	$P + T_i = R + ET_t + T_o + \Delta S$	**(Eq. 5)**

For water quality purposes, we are ultimately interested in the proportion of precipitation that becomes stream/surface flow, also known as *water yield*. Equation 6 describes the relative reduction in water yield from turfgrass with tree canopy relative to turfgrass without canopy, and Eq. 7 describes the relative reduction in water yield from impervious surfaces with tree canopy relative to impervious surfaces without canopy. The subscripts in equations 6 and 7 are used simply to identify the underlying land use (*g* is turfgrass and *i* is impervious) and whether or not that land use is covered by tree canopy (*c*). The role of tree canopy in modifying water yield varies with the severity of precipitation and the quality of vegetation (Keim, Skaugset, and Weiler 2006). To account for spatial and temporal variation in precipitation, we used eleven years (2005 to 2015) of daily weather data (National Climatic Data Center 2016) from each of eight regional locations spanning the Chesapeake Bay Watershed (Figure 1). The ∑ symbol in equations 6 and 7 indicates that the volume of runoff, soil leachate, and change in throughflow will be estimated from daily weather data (unless otherwise noted), and the final results based on the mean annual cumulative total across all sites and years.

$$f_r = \left(1 - \frac{\sum R_{gc} + \sum L_{gc}}{\sum R_g + \sum L_g}\right) \times 100 \qquad \textbf{(Eq. 6)}$$

$$f_r = \left(1 - \frac{\sum R_{ic} + \sum T_{ic}}{\sum R_i + \sum T_i}\right) \times 100 \qquad \textbf{(Eq. 7)}$$

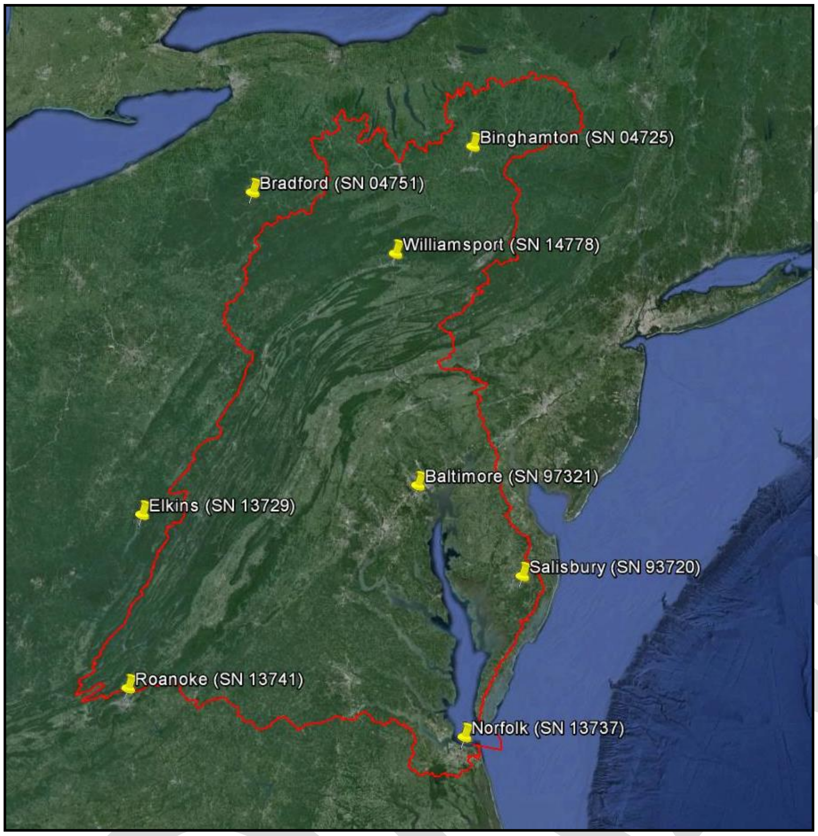


Figure 1: Weather station locations used in this analysis

Precipitation is assumed to be the only input of water to areas mapped as tree canopy over turfgrass. In CBP reporting, riparian forest buffers are a stand alone BMP. This restricts areas mapped as tree canopy over turfgrass to upland sites where the water table is likely below the plant-rooting zone (~2 ft. deep). This assumption likely underestimates the hydrologic benefit of trees in developed areas as trees on residential property have been shown to access water below this depth, uplift, and redistribute it in shallow soil horizons at night (Day, Wiseman et al. 2010). For tree canopy over impervious surfaces, we include a source of shallow, subsurface water (T, throughflow) that trees can access and use to meet basic physiological needs. Throughflow originates from other pervious urban land uses and can be taken up and transpired by trees. The throughflow element is necessary because water and nutrients required for plant growth and function cannot infiltrate through impervious surfaces.

Calculating Runoff

For each of the four land use types, we estimated runoff using the Soil Conservation Service Curve Number Method (Eq. 8), where *R* is runoff, *P* is precipitation, I_a is the initial abstraction, and *S* is the potential maximum retention after runoff begins (all units in inches, USDA 1989).

$$R = \frac{(P - I_a)^2}{(P - I_a) + S} \qquad \textbf{(Eq. 8)}$$

In order to isolate the effects of tree canopy interception beyond the water retaining properties of the underlying land use (i.e., turfgrass and impervious surfaces), we introduced a tree canopy interception term, C_i (units also in inches), into the basic curve number equation (Eq. 9). This term accounts for the amount of precipitation adsorbed to leaves and branches after throughfall stops and, effectively, reduces the amount of precipitation that reaches the ground before initial abstractions occur.

$$R = \frac{(P - C_i - I_a)^2}{(P - C_i - I_a) + S} \qquad \textbf{(Eq. 9)}$$

The amount of precipitation retained in the canopy varies with tree age, species, canopy health, and season. Tree canopy land uses are identified and mapped using satellite imagery with a resolution of one square meter and minimum height of two meters (O'Neil-Dunne et al. 2014), and therefore likely include trees that are ten years old at a minimum. For young to mature deciduous trees, interception capacity ranges from 0.02 to 0.11 inches per storm, whereas interception capacity for coniferous trees of similar age ranges from 0.02 to 0.18 inches per storm (Breuer, Eckhardt, and Frede 2003). For deciduous trees, most but not all interception capacity is lost during the winter as the branches and trunk still provide some interception capacity, whereas conifers retain full interception capacity year round. Because the current mapping methodology only indicates the presence of a tree and provides no other information on tree species or quality of the canopy, C_i in our model was set at 0.05 inches per storm during the growing season (April through October) – below the mean range of interception capacity – and zero during the dormant season. This amount of credit is similar to that given by a state agency for canopy interception in structural BMPs (0.043 in per storm; Minnesota Pollution Control Agency 2016), which likely includes a younger population of trees from new installations rather than a well established tree canopy.

The curve number method was originally designed to estimate stormflow following large precipitation events with flood planning as the most obvious application (Garen and Moore 2005). During the development of this method, observations of precipitation and runoff volume at the watershed scale revealed that the ratio of initial abstractions to potential maximum water retention (I_a/S) was approximately equal to 0.2 (USDA 1989). However, more recent research has demonstrated that this assumption significantly underestimates runoff at smaller scales, and that I_a/S = 0.05 is a more appropriate assumption for small urban areas (Woodward et al. 2003). Substituting 0.05 S for I_a in Equation 9 yields the following:

$$R = \frac{\left(P - C_i - 0.05 \times S\right)^2}{P - C_i + 0.95 \times S} \qquad \textbf{(Eq. 10)}$$

In the curve number method, the total maximum water retaining capacity (S) is further simplified to a dimensionless ‘curve number’ factor, *CN*, ranging from 0 to 100 (Eq. 11). The curve number accounts for the physical attributes of the land surface as well as the hydrologic properties of the underlying soil that affect infiltration. Following recommendations in the USDA Technical Release 55, *Urban Hydrology for Small Watersheds*, we set *CN* = 98 for all impervious surfaces (with and without tree canopy), *CN* = 79 for turfgrass, and *CN* = 74 for tree canopy over turfgrass. The difference between a CN value of 79 and 74 is modest, and is equivalent to turfgrass in fair versus good condition with hydrologic soil group C (HSG). We used a slightly lower curve number (i.e., less runoff) for canopy over turfgrass to account for the *temporal* effects of canopy interception and improved physical structure of compacted soils by tree roots (Day et al. 2010). Equation 12 is the final equation that we used to calculate runoff.

$$S = \frac{1000}{CN} - 10 \qquad \textbf{(Eq. 11)}$$

$$R = \frac{\left(P - C_i - 0.05 \times \left(\frac{1000}{CN} - 10\right)\right)^2}{P - C_i + 0.95 \times \left(\frac{1000}{CN} - 10\right)} \qquad \textbf{(Eq. 12)}$$

Runoff is also influenced by the amount of precipitation and Figure 2 shows how runoff depth varies with precipitation depth based on Equation 12; forested land (*CN* = 55; HSG B) is also shown for reference. As previously mentioned, we used eleven years (2005 to 2015) of daily weather data from each of eight regional locations spanning the Chesapeake Bay Watershed to account for spatial and temporal variation in precipitation.

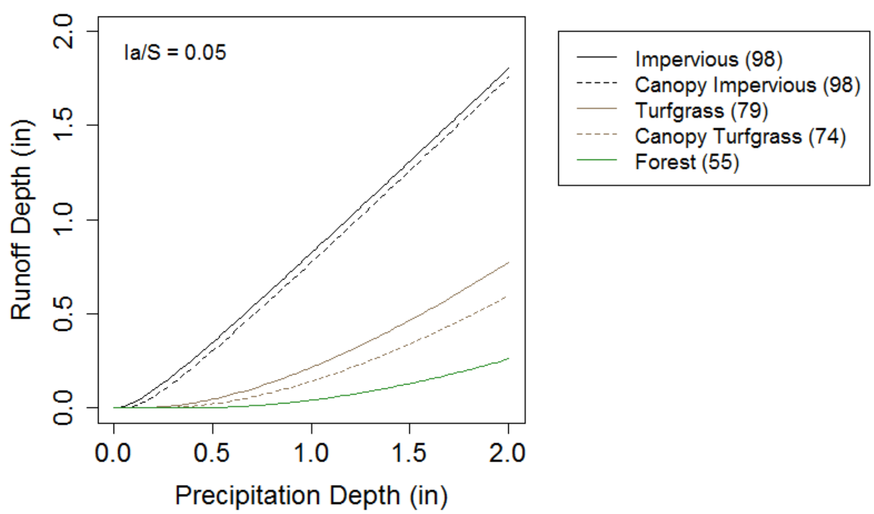


Figure 2: Precipitation vs. runoff depth estimated using the SCS Curve Number Method

For water quality purposes, it is important to note that 'runoff' estimated using the CN method from pervious areas is not the same as 'runoff' from impervious surfaces. In pervious upland areas, runoff includes both infiltration excess overland flow and macro-pore shallow subsurface flow (Garen and Moore 2005). In contrast, runoff from impervious surfaces is exclusively overland flow that is highly connected to surface waters by storm drains and pipes. This implies that on-site retention drives the water quality benefits of tree canopy over turfgrass, where as minimizing downstream erosion drives the water quality benefits of tree canopy over impervious surfaces. While we have no way of quantifying how tree canopy over turfgrass alters the balance between the two components of runoff (i.e., infiltration excess overland flow vs. shallow subsurface flow), it is highly likely that trees reduce infiltration excess overland flow (Asadian and Weiler 2009; Leguédois et al. 2008).

Runoff from impervious and other highly compacted surfaces is particularly problematic in developed areas because water is delivered over shorter periods of time to surface waters that erode stream banks (Walsh, Roy et al. 2005). Water quality can be impaired both by infrequent, large storm events as well as by more frequent, smaller events that deliver quick pulses of nutrients and sediments to receiving waters (Walsh, Fletcher et al. 2005). Trees reduce rainfall intensity and volume, and decrease runoff rates by intercepting precipitation on leaves and branches even when limited by surrounding impervious surfaces (Asadian and Weiler 2009; Nowak, Wang, and Endreny 2007; Wang, Endreny, and Nowak 2008; Xiao et al. 1998).

The results of our runoff calculation are shown in Figure 3. Across all sites and years, annual relative runoff reduction by tree canopy over turfgrass ranged from 18.0 to 39.2 % with a mean value of 29.0 %. Annual runoff reduction by tree canopy over impervious surfaces ranged from 3.5 to 10.6 % with a mean value of 7.0 %. Because sediment does not have a dissolved form, we assumed that sediment retention beneath tree canopy over turfgrass had an efficiency factor of 0.20, resulting in a recommended relative reduction for sediment of 5.8% (29.0% relative reduction in runoff x 0.2).

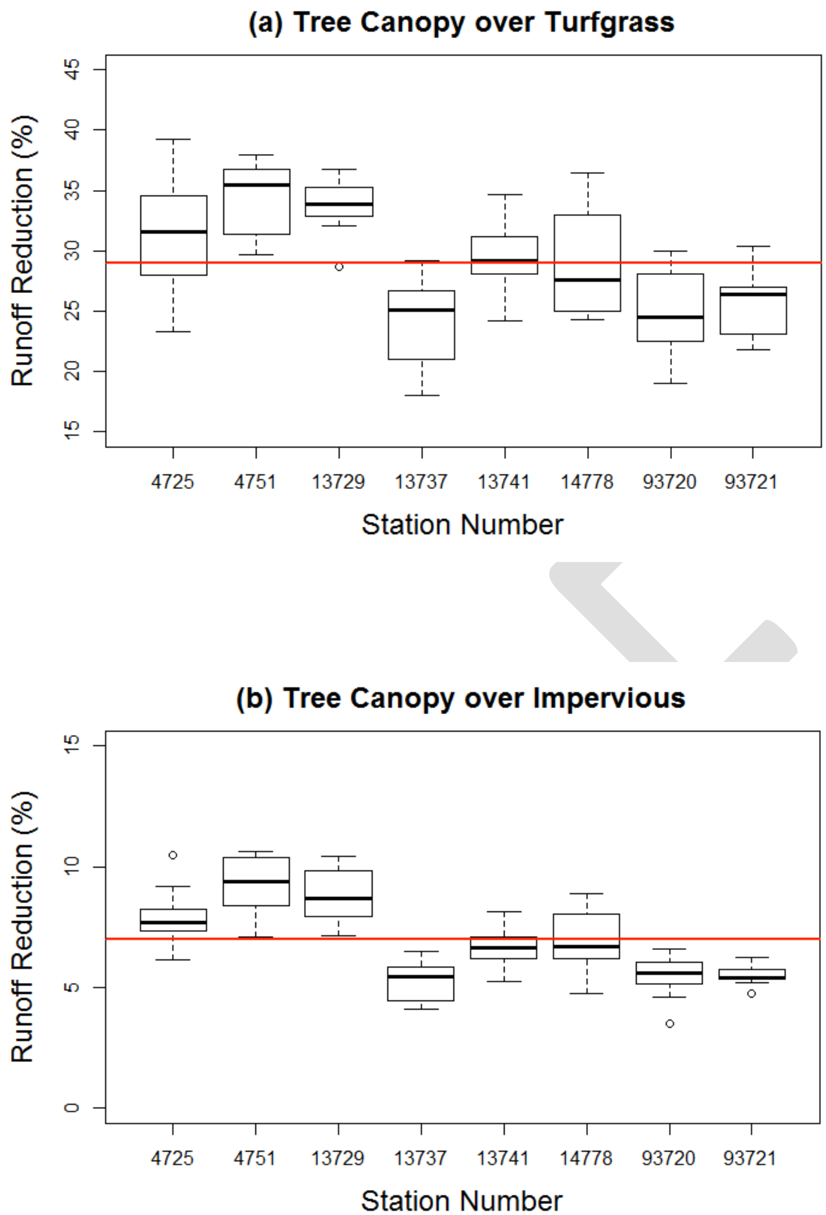


Figure 3: Relative reduction in runoff volume by tree canopy compared to (a) turfgrass and (b) impervious surfaces. Red line indicates the mean relative reduction in runoff across all sites and years.

Calculating Leaching

For pervious urban areas including turfgrass and tree canopy over turfgrass, the precipitation remaining after interception and runoff infiltrates (*I*) into soil (Equation 13). All or a portion of this water is temporarily stored in the soil, and in well-drained areas the maximum amount of soil water storage that is available for evapotranspiration (S_{max}) is equal to the soil's water holding capacity. Water holding capacity varies with soil texture (lowest in both very sandy and very clayey soils) and ranges from ~1 to 2 inches per foot of soil (Brady and Weil 1996). For this analysis, we used a soil water holding capacity of 2 inches per foot, and a total soil volume based on the typical rooting depth of trees (2 ft) and one square meter of land – the minimum mapping unit of tree canopy.

The actual volume of soil water (S_t) will vary over time as a function of the initial soil water volume after rainfall (S_{t-1}) minus the amount of water evaporated and transpired (*ET*) in between precipitation events. Tracking changes in soil water over time also has the advantage of placing an upper limit on ET. During initial model testing, the soil water volume at the end of the year was almost always equal to the maximum water holding capacity. For this reason, the water holding capacity at the beginning of each year was set to zero ($S_t = S_{max}$). Any rainfall that is infiltrated in excess of the available water holding capacity was assumed to leach (L) below the rooting zone and lost as groundwater (Equation 14).

$$I = P - C_i - R \quad \textbf{(Eq. 13)}$$

$$L = I - S_{\max} + S_t \quad \textbf{(Eq. 14)}$$

In between precipitation events, evapotranspiration (ET) by turfgrass and trees reduces the volume of water in soil allowing more infiltrated water to be stored the next time it rains (Equation 15). In the same way that interception varies with tree age, species, canopy health, and season, ET also varies with these factors. Total annual ET rates are similar between turfgrass, trees in natural forest settings, and urban trees ranging from ~15 to 24 inches per year, or 0.04 to 0.064 inches per day (Ford et al. 2011; Penman 1948; Wullschleger et al. 2001; Wullschleger et al. 2000; Wilson et al. 2001; Peters et al. 2010). Again, because the current mapping methodology only indicates the presence of a tree and provides no other information on tree species or quality of the canopy, ET in our model was set at 0.05

inches per day for trees and turfgrass without trees during the growing season (April through October). We assumed that ET of shaded turfgrass was approximately half as effective as non-shaded turfgrass. Therefore, daily rates of ET for turfgrass and canopy over turfgrass were set to 0.05 and 0.08 inches per day during the growing season, respectively. During the dormant season (November through March), ET was only attributed to turfgrass at a rate of 0.025 inches per day.

$$S_t = S_{t-1} - ET \qquad \textbf{(Eq. 15)}$$

The results of our leaching calculation are shown in Figure 4. Across all sites and years, annual relative leaching reduction by tree canopy over turfgrass ranged from 10.0 to 42.5 % with a mean value of 22.5 %.

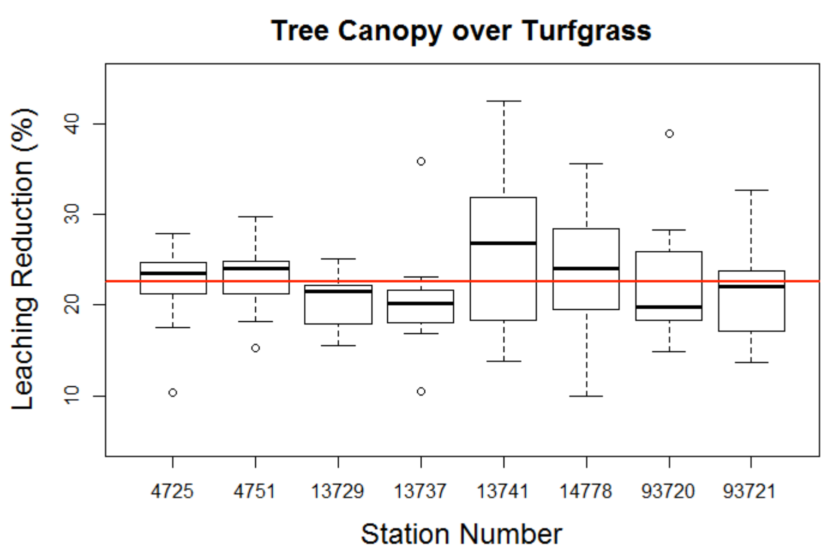


Figure 4: Relative reduction in leaching volume by trees compared to turfgrass

Calculating Throughflow

We know of no straightforward way to estimate the mean daily flux of subsurface throughflow, and subsequent nutrient flux, beneath impervious surfaces. Tree roots are highly advantageous, often growing into broken stormwater pipes and culverts, and there is likely a high degree of spatial variation. In addition, a large portion of dissolved nitrogen and phosphorus taken up through roots with water and incorporated in tree tissues is later deposited as leaf litter on impervious surfaces that are highly connected to surface waters. While it is true that decomposing leaf litter contributes nitrogen and phosphorus to road runoff in autumn (Hobbie et al. 2014; Kaushal and Belt 2012), it is entirely possible that the net flux of N and P in litterfall over the course of a year is zero simply by transforming, concentrating, and redistributing base flow (Janke et al. 2014).

However, a small proportion of N and P are incorporated into woody plant tissues (i.e., branches, bark, heartwood, and sapwood) that represent a long-term store of non-point source pollution. Rather than rely on poorly constrained estimates of throughflow (Eqs. 4 and 5), we choose to estimate relative on-site pollution reduction based on the result of nutrient uptake – biomass growth – and the proportion of N and P stored in wood versus total annual uptake (wood + leaf production) and atmospheric deposition (Eq. 16).

$$R = \left(\frac{W}{W + L + A} \right) \times 100 \qquad \textbf{(Eq. 16)}$$

We estimated annual storage of N and P in tree wood by combining estimates of average annual wood production of urban trees (4.5 to 9.7 kg of C per tree per year, Nowak and Crane 2002) with C, N, and P concentration data in wood from the literature (molar C/N and C/P in wood ~ 300 and 500, respectively; Rastetter et al. 1991; Pettersen 1984; Martin et al. 1998). Based on our calculations (see Appendix 1 for full calculations), 0.038 to 0.083 lbs of N and 0.051 to 0.11 lbs of P are stored in new wood tissues per tree per year. Similarly, we estimated the annual flux of N and P in tree litterfall by combining estimates of annual leaf production (~5 to 20 kg leaf mass per tree per year; Olson 1963; Chapin et al. 2011; Abelho 2001; Martin et al. 1998) with N and P concentration data in tree leaves (~ 25 ppm N and molar N/P ~ 5; Martin et al. 1998; Schaller 1968; McGroddy et al. 2004). These calculations indicate that ~ 0.27 to 1.10 lbs of N and 0.12 to 0.49 lbs of P are taken up each year by trees and used to build leaves. Finally, we used established atmospheric deposition rates (14 kg N ha^{-1} yr^{-1}; National Atmospheric Deposition Program) or precipitation concentration data combined with annual rainfall data (63 ppb inorganic + organic P ; Smullen et al. 1982) to estimate the amount of atmospheric N and P deposited on the canopy of a mature tree (~ 10 m^2). Table 1 shows the fluxes of N and P used to estimate storage in wood relative to total uptake and atmospheric deposition.

Table 1. Average annual nutrient uptake and atmospheric deposition of N and P

Elemental Fluxes	Woody Biomass (lbs. $tree^{-1}$ yr^{-1})	Litterfall (lbs. $tree^{-1}$ yr^{-1})	Atmos. Dep. (lbs. $tree^{-1}$ yr^{-1})	N and P in wood relative to total uptake and atm. Dep. (%)
Nitrogen	0.038	0.69	0.031	5.0 %
Phosphorus	0.051	0.31	0.0013	14 %

To make these results comparable to water yield, we must still make some assumption about the efficiency of nutrient uptake over the course of a year. For this conversion we used a simple assumption based on the proportion of time that trees are transpiring over the course of a year (7/12 months x ½ day = 0.29). Therefore, we estimate that the relative reduction in N and P by trees from throughflow to be 1.5 and 4.0 % respectively.

Summary of results and recommendations

The results of our water yield calculations are summarized in Table 2. Relative reduction in total yield for tree canopy over turfgrass is closer to the relative reduction in leaching due to the greater volume of water exported in soil leachates compared to runoff. Table 3 displays the final recommended relative reductions for N, P, and sediment for tree canopy land uses after adding in the small proportion of N and P stored in woody biomass.

Table 2. Estimated annual reductions in water yield by tree canopy relative to impervious and pervious land covers

Land Use	Precip. (in)	Runoff Red. (%)	Leaching Red. (%)	Total Yield (%)
Canopy over Turfgrass	39.9	29.0	22.5	23.8
Canopy over Impervious	39.9	7.0	NA	7.0

Table 3. Recommended relative reductions in N, P, and sediment for tree canopy land uses

Land Use	Total Nitrogen Red. (%)	Total Phos. Red. (%)	Total Sed. Red. (%)
Canopy over Turfgrass	23.8	23.8	5.8
Canopy over Impervious	8.5	11.0	7.0

References

Abelho, Manuela. 2001. “From Litterfall to Breakdown in Streams: A Review.” *The Scientific World Journal* 1: 656–80.

Asadian, Yeganeh, and Markus Weiler. 2009. “A New Approach in Measuring Rainfall Interception by Urban Trees in Coastal British Columbia.” *Water Quality Research Journal of Canada* 44 (1): 16.

Brady, N. C., and R. R. Weil. 1996. “The Nature and Properties of Soils.,” no. Ed. 11: xi + 740 pp.

Breuer, Lutz, Klaus Eckhardt, and Hans-Georg Frede. 2003. “Plant Parameter Values for Models in Temperate Climates.” *Ecological Modelling* 169 (2–3): 237–93. doi:10.1016/S0304-3800(03)00274-6.

Campbell, John L., James W. Hornbeck, Myron J. Mitchell, Mary Beth Adams, Mark S. Castro, Charles T. Driscoll, Jeffrey S. Kahl, et al. 2004. “Input-Output Budgets of Inorganic Nitrogen for 24 Forest Watersheds in the Northeastern United States: A Review.” *Water, Air, and Soil Pollution* 151 (1-4): 373–96.

Chapin III, F. Stuart, Pamela A. Matson, and Peter Vitousek. 2011. *Principles of Terrestrial Ecosystem Ecology*. Springer Science & Business Media. https://books.google.com/books?hl=en&lr=&id=68nFNpceRmIC&oi=fnd&pg=PR5&dq=principles+of+terrestrial+ecosystem+ecology&ots=V1FWdx8qrl&sig=gyLoKQiwmdhAJaMcLHFsULCVABs.

Cole, DALE W., and STANLEY P. Gessel. 1992. “Fundamentals of Tree Nutrition.” *Forest Fertilization: Sustaining and Improving Nutrition and Growth of Western Forests. Seattle: Institute of Forest Resources Contribution*, no. 73: 7–16.

Correll, David L. 1998. “The Role of Phosphorus in the Eutrophication of Receiving Waters: A Review.” *Journal of Environmental Quality* 27 (2): 261–66.

Daniels, W. Lee, Mike Goatley, Rory Maguire, and David Sample. 2010. “Effects of Fertilizer Management Practices on Urban Runoff Water Quality.” https://www.researchgate.net/profile/David_Sample/publication/228869182_Effects_of_Fertilizer_Management_Practices_on_Urban_Runoff_Water_Quality/links/0046351b313bc740a7000000.pdf.

Day, Susan D., P. Eric Wiseman, Sarah B. Dickinson, and J. Roger Harris. 2010. “Tree Root Ecology in the Urban Environment and Implications for a Sustainable Rhizosphere.” *Journal of Arboriculture* 36 (5): 193.

Ford, Chelcy R., Robert M. Hubbard, and James M. Vose. 2011. “Quantifying Structural and Physiological Controls on Variation in Canopy Transpiration among Planted Pine and Hardwood Species in the Southern Appalachians.” *Ecohydrology* 4 (2): 183–95.

Garen, David C., and Daniel S. Moore. 2005. *CURVE NUMBER HYDROLOGY IN WATER QUALITY MODELING: USES, ABUSES, AND FUTURE DIRECTIONS1*. Wiley Online Library. http://onlinelibrary.wiley.com/doi/10.1111/j.1752-1688.2005.tb03742.x/full.

Gift, Danielle M., Peter M. Groffman, Sujay S. Kaushal, and Paul M. Mayer. 2010. “Denitrification Potential, Root Biomass, and Organic Matter in Degraded and Restored Urban Riparian Zones.” *Restoration Ecology* 18 (1): 113–20.

Groffman, Peter M., Neely L. Law, Kenneth T. Belt, Lawrence E. Band, and Gary T. Fisher. 2004. “Nitrogen Fluxes and Retention in Urban Watershed Ecosystems.” *Ecosystems* 7 (4): 393–403.

Hobbie, Sarah E., Lawrence A. Baker, Christopher Buyarski, Daniel Nidzgorski, and Jacques C. Finlay. 2014. “Decomposition of Tree Leaf Litter on Pavement: Implications for Urban Water Quality.” *Urban Ecosystems* 17 (2): 369–85.

Huyler, Ann, Arthur H. Chappelka, Stephen A. Prior, and Greg L. Somers. 2014. “Influence of Aboveground Tree Biomass, Home Age, and Yard Maintenance on Soil Carbon Levels in Residential Yards.” *Urban Ecosystems* 17 (3): 787–805. doi:10.1007/s11252-014-0350-7.

Janke, Benjamin D., Jacques C. Finlay, Sarah E. Hobbie, Larry A. Baker, Robert W. Sterner, Daniel Nidzgorski, and Bruce N. Wilson. 2014. “Contrasting Influences of Stormflow and Baseflow Pathways on Nitrogen and Phosphorus Export from an Urban Watershed.” *Biogeochemistry* 121 (1): 209–28.

Jo, Hyun-Kil, and Gregory E. McPherson. 1995. “Carbon Storage and Flux in Urban Residential Greenspace.” *Journal of Environmental Management* 45 (2): 109–33.

Kaushal, Sujay S., and Kenneth T. Belt. 2012. “The Urban Watershed Continuum: Evolving Spatial and Temporal Dimensions.” *Urban Ecosystems* 15 (2): 409–35.

Kaushal, Sujay S., Peter M. Groffman, Paul M. Mayer, Elise Striz, and Arthur J. Gold. 2008. “Effects of Stream Restoration on Denitrification in an Urbanizing Watershed.” *Ecological Applications* 18 (3): 789–804.

Keim, R. F., A. E. Skaugset, and M. Weiler. 2006. “Storage of Water on Vegetation under Simulated Rainfall of Varying Intensity.” *Advances in Water Resources* 29 (7): 974–86. doi:10.1016/j.advwatres.2005.07.017.

Leguédois, Sophie, Tim W. Ellis, Peter B. Hairsine, and David J. Tongway. 2008. “Sediment Trapping by a Tree Belt: Processes and Consequences for Sediment Delivery.” *Hydrological Processes* 22 (17): 3523–34.

Martin, Jonathan G., Brian D. Kloeppel, Tara L. Schaefer, Darrin L. Kimbler, and Steven G. McNulty. 1998. “Aboveground Biomass and Nitrogen Allocation of Ten Deciduous Southern Appalachian Tree Species.” *Canadian Journal of Forest Research* 28 (11): 1648–59.

McGroddy, Megan E., Tanguy Daufresne, and Lars O. Hedin. 2004. “Scaling of C: N: P Stoichiometry in Forests Worldwide: Implications of Terrestrial Redfield-Type Ratios.” *Ecology* 85 (9): 2390–2401.

McHale, M. R., I. C. Burke, M. A. Lefsky, P. J. Peper, and E. G. McPherson. 2009. “Urban Forest Biomass Estimates: Is It Important to Use Allometric Relationships Developed Specifically for Urban Trees?” *Urban Ecosystems* 12 (1): 95–113.

Minnesota Pollution Control Agency. 2016. “Calculating Credits for Tree Trenches and Boxes.” *Minnesota Stormwater Manual*. http://stormwater.pca.state.mn.us/index.php/Calculating_credits_for_tree_trenches_and_tree_boxes.

National Climatic Data Center. 2016. “Quality Controlled Local Climatological Data.” http://www.ncdc.noaa.gov/qclcd/QCLCD?prior=N.

Nowak, David J., and Daniel E. Crane. 2002. “Carbon Storage and Sequestration by Urban Trees in the USA.” *Environmental Pollution* 116 (3): 381–89.

Nowak, David J., Jun Wang, and Ted Endreny. 2007. “Environmental and Economic Benefits of Preserving Forests within Urban Areas: Air and Water Quality.” *The Economic Benefits of Land Conservation*, 29.

Olson, Jerry S. 1963. “Energy Storage and the Balance of Producers and Decomposers in Ecological Systems.” *Ecology* 44 (2): 322–31.

O’Neil-Dunne, Jarlath, Sean MacFaden, Anna Royar, Maxwell Reis, Ralph Dubayah, and Anu Swatantran. 2014. “An Object-Based Approach to Statewide Land Cover Mapping.” In *Proceedings of ASPRS 2014 Annual Conference*, 23–28. http://www.asprs.org/a/publications/proceedings/Louisville2014/ONeilDunne.pdf.

Penman, Howard Latimer. 1948. “Natural Evaporation from Open Water, Bare Soil and Grass.” In *Proceedings of the Royal Society of London A: Mathematical, Physical and Engineering Sciences*, 193:120–45. The Royal Society. http://rspa.royalsocietypublishing.org/content/royprsa/193/1032/120.full.pdf.

Peters, Emily B., Joseph P. McFadden, and Rebecca A. Montgomery. 2010. “Biological and Environmental Controls on Tree Transpiration in a Suburban Landscape.” *Journal of Geophysical Research: Biogeosciences* 115 (G4). http://onlinelibrary.wiley.com/doi/10.1029/2009JG001266/full.

Raciti, Steve M., Amy J. Burgin, Peter M. Groffman, David N. Lewis, and Timothy J. Fahey. 2011. “Denitrification in Suburban Lawn Soils.” *Journal of Environmental Quality* 40 (6): 1932–40.

Rastetter, Edward B., Michael G. Ryan, Gaius R. Shaver, Jerry M. Melillo, Knute J. Nadelhoffer, John E. Hobbie, and John D. Aber. 1991. “A General Biogeochemical Model Describing the Responses of the C and N Cycles in Terrestrial Ecosystems to Changes in CO2, Climate, and N Deposition.” *Tree Physiology* 9 (1-2): 101–26.

ROGER C. PETTERSEN. 1984. “The Chemical Composition of Wood.” In *The Chemistry of Solid Wood*, 207:57–126. Advances in Chemistry 207. American Chemical Society. http://dx.doi.org/10.1021/ba-1984-0207.ch002.

Schaller, Friedrich. 1968. “Soil Animals.” University of Michigan Press Ann Arbor. http://books.google.com/books/about/Soil_animals.html?id=H1tIAAAAMAAJ.

Smullen, James T., J. L. Taft, and Joseph Macknis. 1982. “Nutrient and Sediment Loads to the Tidal Chesapeake Bay System.” *United States Environmental Protection Agency, Chesapeake Bay Program*, 147–258.

Staver, Kenneth W., and Russell B. Brinsfield. 2001. “Agriculture and Water Quality on the Maryland Eastern Shore: Where Do We Go from Here? Long-Term Solutions to Accelerated Eutrophication Must Provide Mechanisms for Redistributing Nutrients Flowing into Concentrated Animal-Producing Regions.” *BioScience* 51 (10): 859–68. doi:10.1641/0006-3568(2001)051[0859:AAWQOT]2.0.CO;2.

Takahashi, Terumasa, Yoshihiro Amano, Kayo Kuchimura, and Tatsuaki Kobayashi. 2008. “Carbon Content of Soil in Urban Parks in Tokyo, Japan.” *Landscape and Ecological Engineering* 4 (2): 139–42.

USDA, Natural Resources Conservation Service, Conservation Enginering Divisions. 1989. “Urban Hydrology for Small Watersheds.” *TR-55*.

Vannote, Robin L., G. Wayne Minshall, Kenneth W. Cummins, James R. Sedell, and Colbert E. Cushing. 1980. “The River Continuum Concept.” *Canadian Journal of Fisheries and Aquatic Sciences* 37 (1): 130–37. doi:10.1139/f80-017.

Wakida, Fernando T., and David N. Lerner. 2005. “Non-Agricultural Sources of Groundwater Nitrate: A Review and Case Study.” *Water Research* 39 (1): 3–16.

Wang, Jun, Theodore A. Endreny, and David J. Nowak. 2008. *Mechanistic Simulation of Tree Effects in an Urban Water Balance Model1*. Wiley Online Library. http://onlinelibrary.wiley.com/doi/10.1111/j.1752-1688.2007.00139.x/full.

Wilson, Kell B., Paul J. Hanson, Patrick J. Mulholland, Dennis D. Baldocchi, and Stan D. Wullschleger. 2001. “A Comparison of Methods for Determining Forest Evapotranspiration and Its Components: Sap-Flow, Soil Water Budget, Eddy Covariance and Catchment Water Balance.” *Agricultural and Forest Meteorology* 106 (2): 153–68.

Woodward, Donald E., Richard H. Hawkins, Ruiyun Jiang, Allen T. Hjelmfelt, Jr., Joseph A. Van Mullem, and Quan D. Quan. 2003. “Runoff Curve Number Method: Examination of the Initial Abstraction Ratio.” In , 1–10. American Society of Civil Engineers. doi:10.1061/40685(2003)308.

Wullschleger, Stan D., P. J. Hanson, and D. E. Todd. 2001. “Transpiration from a Multi-Species Deciduous Forest as Estimated by Xylem Sap Flow Techniques.” *Forest Ecology and Management* 143 (1): 205–13.

Wullschleger, Stan D., Kell B. Wilson, and Paul J. Hanson. 2000. “Environmental Control of Whole-Plant Transpiration, Canopy Conductance and Estimates of the Decoupling Coefficient for Large Red Maple Trees.” *Agricultural and Forest Meteorology* 104 (2): 157–68.

Xiao, Qingfu, E. Gregory McPherson, James R. Simpson, and Susan L. Ustin. 1998. “Rainfall Interception by Sacramento’s Urban Forest.” *Journal of Arboriculture* 24: 235–44.

Zhu, Wei-Xing, Noah D. Dillard, and Nancy B. Grimm. 2005. “Urban Nitrogen Biogeochemistry: Status and Processes in Green Retention Basins.” *Biogeochemistry* 71 (2): 177–96.

Appendix 2: Biomass Calculations

Woody Biomass

4.5 to 9.7 kg of C per tree per year (Nowak and Crane 2002)

x 1000 g/kg conversion factor

x 1 mol C / 12.01 g of C conversion factor

x 1 mol N / 300 mol C references listed below

x 14.01 g N / mol N

x 1 lb / 454 g

= 0.038 to 0.083 lbs N per tree per year

mass ratio C/N approximately 300:1 (Rastetter et al. 1991), C is 50% of wood (so 500 mg/g) (Pettersen 1984), 0.2 to 4.0 mg N / g wood for heartwood, sapwood, and branches (Martin et al. 1998)

4.5 to 9.7 kg of C per tree per year (Nowak and Crane 2002)

x 1000 g/kg conversion factor

x 1 mol C / 12.01 g of C

x 1 mol P / 500 mol C (ROGER C. PETTERSEN 1984)

x 30.97 g P / mol P

x 1 lb / 454 g

= 0.051 to 0.11 lbs P per tree per year

Litterfall

5 to 20 kg per tree per yr (Olson 1963; Chapin III, Matson, and Vitousek 2011; Abelho 2001) (Martin et al. 1998)

x 1000 g / kg unit conversion factor

x 25 mg N / 1 g of litter (Martin et al. 1998)

x 1 g / 1000 mg unit conversion factor

x 1 lb / 454 g unit conversion factor

= 0.27 to 1.10 lbs N per tree per year

5 to 20 kg per tree per yr (Olson 1963; Chapin III, Matson, and Vitousek 2011; Abelho 2001) (Martin et al. 1998)

x 1000 g / kg unit conversion factor

x 25 mg N / 1 g of litter (Martin et al. 1998; Schaller 1968)

x 1 g / 1000 mg unit conversion factor

x 1 mol N / 14.01 g N unit conversion factor

x 1 mol P / 5 mol N (McGroddy, Daufresne, and Hedin 2004)

x 30.97 g P / mol P unit conversion factor

x 1 lb / 454 g unit conversion factor

= 0.12 to 0.49 lbs P per tree per year

Atmospheric Deposition

14 kg N ha^{-1} yr^{-1} (NADP via Marion)

x 1000 g / 1 kg unit conversion factor

x 1 lb. / 454 g unit conversion factor

x 1 ha / 10000 m^2 unit conversion factor

x 10 m^2 canopy area of single tree

= 0.031 lb. N per tree per year

Appendix 2, continued

Atmospheric Deposition

TP = 63 ug/L in atmospheric dep. (Smullen, Taft, and Macknis 1982)

x 1 g / 10^6 ug

x 1 lb. / 454 g

x 1000 L / m^3

x 39.9 in of rainfall

x 1 m / 39.37 in

x 10 m^2

= 0.0013 lbs P per tree per year

Appendix C: Urban Tree Canopy Literature Synthesis

Review of the Available Science for Urban Tree Planting and Canopy

A total of 115 publications were reviewed by the Expert Panel to evaluate the research questions defined in the scope of this Expert Panel:

1. What is the effectiveness of urban tree canopy on reducing runoff, nutrient and sediment?
2. How does effectiveness vary by species, over time, with differences in planting sites (e.g., distance from impervious cover or other trees, soil conditions, geographic location) and with different maintenance strategies?

A limited number of studies directly address the water quality benefits of urban trees, and an even smaller subset provide results that can be used to develop effectiveness values for urban tree planting. Consequently, the data reviewed were not limited to the Chesapeake Bay watershed. Of greater applicability were the 49 studies on the hydrologic benefits of urban trees. These studies attempt to quantify one or more components of the tree's hydrologic cycle, which, combined, can inform estimates of runoff reduction provided by urban trees. The literature was also extended to include studies on trees planted as a part of an urban stormwater best management practice to quantify the impact of urban tree canopy on water quality. We also reviewed a number of studies on the water quality and runoff reduction benefits of non-urban forests, which may be considered an upper limit to any credit assigned to urban tree planting, based on the assumption that trees and forests in urban environments do not function as well as natural forests due to factors such as compacted soils, lack of understory, open-grown trees and numerous impacts on tree health.

Because trees planted in the urban riparian zone (i.e., within 100 feet of a waterbody) will be credited under a separate best management practice (Urban Forest Buffers), this review focused primarily on the benefits of trees in upland areas. Urban trees provide a host of other benefits, including air quality improvement, habitat for wildlife, temperature reduction and energy savings. While some of these ancillary benefits were also addressed in the literature reviewed, this synthesis focuses solely on nutrient, sediment and runoff reduction.

Hydrologic Benefits

Trees affect water quality primarily by reducing the amount of stormwater runoff that reaches surface waters. Trees reduce runoff through rainfall interception by the tree canopy, by releasing water into the atmosphere through evapotranspiration (ET), and by promoting infiltration of water through the soil and storage of water in the soil and forest litter. Major findings from the literature review for each of these processes are summarized below.

Interception

Canopy interception of rainfall is an important and significant component of the tree water balance. Interception losses depend on factors such as leaf area index (LAI), tree structure, and notably storm characteristics (Xiao et al. 2000). The most critical time for trees to play a role in reducing runoff is during and right after a storm (KDGT 2013). KDGT (2013) suggests that, because of this, continuous simulation modeling may be the best approach for estimating rainfall interception on an annual basis. Quantifying interception on an annual timescale allows for development of an average value across differing rainfall conditions and also accounts for interception during both leaf-on and leaf-off periods. Therefore, the synthesis of studies in this section focuses primarily on those reporting results over an annual timeframe.

Table 1 summarizes the values found in the literature on annual rainfall interception by urban trees and forests, which range from 6.5 to 66.5% for all trees, 6.5 to 27% for deciduous trees and 27-66% for evergreen species, as a percent of annual rainfall. Some of the studies only reported interception as a volume per tree per year and these results range from 106 to 2,000 gallons/tree/year. Note that most of the studies in Table 1 are from semi-arid climates, so further analysis will be needed to adapt them to the Chesapeake Bay region.

More studies are available on rainfall interception by natural forests, and these results are summarized in Table 2 for comparison to the urban tree results. Even in the natural environment, rainfall interception by forests is extremely variable and difficult to measure, as noted by Crockford and Richardson (2000) in a review of interception studies. The range of annual interception by deciduous forests shown in Table 2 is 10-22% and 15-46% for evergreen forests. Both sets of data generally agree that evergreen intercept more rainwater than deciduous trees (more than double in some cases) since they have leaves year-round.

Table 1. Rainfall Interception Studies of Urban Trees

Study	Location	Interception (% of annual rainfall)[1]	Species/Condition[2]	Type of Study[3]
Kirnbauer et al. 2013	Hamilton, Ontario, CA	6.5-11 17-27	G. biloba (D), P. acerifolia (D), A. saccharinum (D) L. styraciflua (D)	Modeling
Livesley et al. 2014	Melbourne, Victoria, Aus.	29 44	E. saligna (E) E. nicholii (E)	Measured
Xiao and McPherson 2002	Santa Monica, CA	27.3 15.3 66.5	All park and street trees Small jacaranda mimosifolia (D) Mature tristania conferta (E)	Modeling
Xiao et al. 1998	Sacramento County, CA	11.1	Tree canopy in the County	Modeling
Xiao et al. 2000	Davis, CA	15 27	Pear (D) Oak (E)	Measured
Xiao and McPherson 2011a	Oakland, CA	14.3 25.2 27.0	Sweetgum (D) Gingko (D) Lemon (E)	Measured
Wang et al. 2008	Baltimore, MD	18.4	Tree canopy in Dead Run subwatershed (D)	Modeling
Band et al. 2010	Fairfax, VA	14.5	Tree canopy in Accotink watershed (D)	Modeling
Band et al. 2010	Baltimore, MD	15.7	Tree canopy in Gwynns Falls watershed (D)	Modeling
Band et al. 2010	Montgomery County, MD	19.6	Tree canopy in Rock Creek watershed (D)	Modeling
Asadian and Weiler (2009)	Vancouver, BC	49 61	Douglas fir (E) Western red cedar (E)	Measured
Study	Location	Interception	Species/Condition	Type of

Table 1. Rainfall Interception Studies of Urban Trees

		(gallons/ tree/yr)[4]		Study
Berland and Hopton 2014	Cincinnati, OH	1,770 (6.7)	Average value	Modeling
McPherson and Simpson 2002	Modesto, CA	845 (3.2)	Average value	Modeling
McPherson and Simpson 2002	Santa Monica, CA	1,849 (7.0)	Average value	Modeling
McPherson et al. 2011	Los Angeles, CA	106 (0.4) (low) 1,479 (5.6 (high)	Crapemyrtle Jacaranda (D)	Modeling
Soares et al. 2011	Lisbon, Portugal	1,189 (4.5)	Average value	Modeling
CWP, 2014	Montgomery County, MD	2,000 (7.57)	15-20 year old 9-15” DBH tree	Modeling

[1] represents the % of rain falling on the tree canopy that is captured through interception
[2] D = deciduous, E = evergreen
[3] Measured = studies that infer interception by subtracting measured throughflow and stemflow from measured rainfall; modeled = studies that model interception using models such as i-tree
[4] Units of m^3/tree/yr are noted in parentheses

Table 2. Rainfall Interception by Natural Forests

Study	Interception (% of annual rainfall)	Type of Forest/Location	Type of Study
Zinke (1967), cited in Xiao et al. (2000)	15-40	Conifer stands	Compilation of 39 Studies
	10-20	Hardwood stands	
Baldwin (1938), cited in Xiao et al. (2000)	59	Old growth forests	Unknown
Dunne and Leopold (1978) cited in Herrera Environmental Consultants (2008)	13[1]	Deciduous trees	Compilation of measured studies
	28[1]	Conifers	
Molchanov (1960) cited in Reynolds et al. (1988)	34-46	Spruce forest/USSR	Measured
	24-27	Pine forest/USSR	
	24	Birch forest/USSR	
	22	Oak forest/USSR	
Heal et al. (2004)	44	Conifers/UK	Measured
Link et al. (2004)	22.8-25	Old-growth Douglas fir forest/Western Cascades, WA	Measured
Deguchi et al. (2006)	16.8	Deciduous forest/Japan	Measured

[1] these studies were unavailable so it is unknown whether these values represent percent of annual rainfall versus storm event or study period rainfall

Evapotranspiration

Evaporation and transpiration occur simultaneously and there is no easy way of distinguishing between the two processes. When vegetation is small, water is predominately lost by soil evaporation, but once the vegetation is well developed, transpiration becomes the main process. As described in KDGT (2013), rainfall interception, advection, turbulent transport, total leaf surface area and available water capacity are all factors that combine to control ET rates, and the relative importance of each variable can fluctuate due to climate, soils and vegetative conditions.

Given the complexity of quantifying ET, no studies were found that quantify annual ET rates for trees in urban areas. Most studies instead evaluate how one or more factors influence ET, develop and test models for estimating ET, or measure ET values for a particular species during the growing season. KDGT (2013) describe the different methods of estimating ET, as well as the advantages and limitations of each.

Sinclair et al. (2005) documented the influence of soil moisture on ET and found that ET is highest when soil moisture is highest, and decreases as soil moisture decreases. Wang et al. (2011) found that transpiration rates were highest during a summer day and lowest during a winter night because of the great influence of the evaporative demand index, consisting of air temperature, soil temperature, total radiation, vapor pressure deficit, and atmospheric ozone. Guidi et al (2008) concluded that ET was strongly correlated to plant development and mainly dependent on its nutritional status rather than on the differences between species. A modeling study by Band et al. (2010) in suburban watershed in Baltimore County, MD, identified the importance of ET on runoff reduction and noted that the major effect of tree canopy on runoff production was the ability to remove soil water by transpiration, allowing more pore space for infiltration. However, Litvak et al (2014) found that in summer, total plot ET of urban lawns with trees was lower than lawns without trees by 0.9–3.9 mm d^{-1} in the Los Angeles metropolitan area. Another study from Los Angeles by Pataki et. al (2011) raised concerns that certain tree species may place too much of a demand on the local water supply because of high ET rates.

Tables 3 and 4 present a summary of transpiration studies for urban trees while Table 5 summarizes similar data from natural forests. These studies quantify transpiration during the growing season or some portion of it, rather than on an annual basis. There is quite a wide range of results for the average daily volume of water an urban tree can transpire, from 0.2 gallons to 46.7 gallons per tree per day. Studies that report rates of transpiration show a more narrow range of results, from 0.1 to 2.39 mm/day for urban trees. These rates are comparable to that of natural forests, which range from 0.5 to 3.0 mm/day.

Table 3. Transpiration Rates by Urban Trees During the Growing Season				
Study	**Location**	**Average Daily Transpiration Rate (mm/day)**	**Species / Condition[1]**	**Type of Study**
Wang (2012)	Beijing, China	1.47	Horse Chestnut - *Aesculus chinensis* (D), 10.5-19.2 DBH	Measured
Chen et al. (2011)	Liaoning Province, China	1.51	*Cedrus deodara, Zelkova schneideriana, Metasequoia glyptostroboides, Euonymus bungeanus*	Measured
Peters et al. (2010)	Minneapolis St. Paul, Minnesota	1.1[2]	*Fraxinus Pennsylvanica, Quercus rubra, Juglans nigra, Tilia Americana, Ulmus pumila, Ulmus thomasii* (D)	Measured
		1.9[2]	*Picea glauca, Picea pungens, Pinus strobes, Picea abies, Pinus nigra, Pinus sylvestris* (E)	Measured
Cermak et al. (2000)	City of Brno, Czech Republic	2.17	Red Maple - *Acer campestre L* (D), roots covered by asphalt, 18” DBH, shaded	Measured
		2.39	Red Maple - *Acer campestre L* (D), roots covered by asphalt, 50” DBH, exposed to sunlight	
Pataki et a. (2011)	Los Angeles, CA	0.1-2.2	Urban forest plots with mixed species	Measured

[1]D = deciduous, E = evergreen

[2] Converted from $kg/m^2/day$ assuming 1kg = $0.0010m^3$

Table 4. Gallons of Water Transpired by Urban Trees During the Growing Season				
Study	**Location**	**Average Daily Transpiration Volume (gal/tree/day)**	**Species / Condition[2]**	**Type of Study**
Pataki et al. (2011)	Los Angeles, CA	0.2[3]	Laurel Sumac - *Malosma laurina*, unirrigated	Measured
		0.8[3]	*Pinus canariensis*, unirrigated	
		2.3[3]	Blue Jacaranda - *Jacaranda mimosifolia*, irrigated	

Table 4. Gallons of Water Transpired by Urban Trees During the Growing Season				
Study	**Location**	**Average Daily Transpiration Volume (gal/tree/day)**	**Species / Condition[2]**	**Type of Study**
		3.4[3]	Kurrajong - *Brachychiton populneus*	
		3.4[3]	Redwood - *Sequoia sempervirens*	
		5.0[3]	Lacebark –*Brachychiton discolor*	
		11.3[3]	Grand Eucalyptus - *Eucalyptus grandis*	
		12.0[3]	Crape Myrtle - *Lagerstroemia indica*	
		12.5[3]	California Sycamore - *Platanus racemosa*, campus	
		13.0[3]	Canary Island Pine - *Pinus canariensis*,	
		13.4[3]	Goldenrain tree - *Koelreuteria paniculata*	
		17.9[3]	Chinese elm - *Ulmus parvifolia*	
		19.4[3]	*Pinus canariensis*, campus	
		23.7[3]	Laurel Fig - *Ficus microcarpa*	
		23.7[3]	Honey Locust - *Gleditsia triacanthos*	
		26.2[3]	Jacaranda - *Jacaranda chelonia*	
		27.1[3]	*Platanus racemosa*, street	
		46.7[3]	London Planetree - *Platanus hybrida*, street	
Green (1993)	Palmerston North, New Zealand	10.5[1]	10 year old isolated walnut (D)	Measured
Cermak et al. (2000)	City of Brno, Czech Republic	17[1]	Red Maple - *Acer campestre L* (D), roots covered by asphalt, 18” DBH, shaded	Measured
		37[1]	Red Maple - *Acer campestre L* (D), roots covered by asphalt, 50” DBH, exposed to sunlight	

[1] Converted from liters/tree/day

[2]D = deciduous, E = evergreen

[3]Converted from kg/tree/day assuming 1 gallon = 3.79 kg of water

Table 5. Transpiration Rates by Natural Forests During the Growing Season

Study	Location	Average Daily Transpiration Rate (mm/day)	Type of Forest/Location	Type of Study
Wullschleger et al. (2000)	Eastern TN	1.1-3.0[1]	Large red maples in a upland oak forest	Measured
Wullschleger et al. (2001)	Eastern TN	1.1 (average) 2.2 (maximum)	Upland oak forest (white and red oak, black gum, red maple, yellow poplar)	Measured / Modeled[2]
Cienciala et al. (1997)	Central Sweden	0.5[3]	100 year old stand sub-boreal forest (pine and spruce)	Measured
		0.9[3]	50 year old stand sub-boreal forest (pine and spruce)	
Ford et al. (2011)	Coweeta Basin, Western NC	1.1	Mixed deciduous hardwood forest	Measured
		2.4	White pine forest	

[1]Measurements are for individual trees

[2]Sap flow measurements for individual trees were used to model stand transpiration

[3]Measurements taken during a dry period in July

Because of the difficulty in measuring ET by trees over annual timeframes, some studies use a water balance approach to estimate ET for a watershed by subtracting discharge from precipitation. Table 6 summarizes these studies, which estimated annual ET rates for forested watersheds of 24% to 77%. Hibbert (1969) found that water yield from a 22-acre catchment in the southern Appalachians increased over 5 inches annually when the catchment was converted from hardwood forest to grass. During years when grass production was high, water yield from the catchment was about the same as or less than the expected yield from the original forest. As grass productivity declined, water yield gradually increased. Hibbert (1969) attributes the changes in water yield to changes in ET.

Table 6. Annual ET by Natural Forests

Study	Location	Results	Description
Boggs and Sun (2011)	Central North Carolina piedmont	Forested watershed retained 77% of annual rainfall, compared to 58% for an urban watershed with 44% impervious cover	ET was estimated by subtracting measured streamflow from precipitation.
Post and Jones (2001)	Oregon, New Hampshire, North Carolina and Puerto Rico	Deciduous forested basins retain 24-54% of rainfall and evergreen forests retain 43-50% of rainfall	ET was estimated by subtracting measured streamflow from precipitation.

Infiltration

Studies on the effects of urban trees on soil infiltration are limited. The studies reviewed demonstrate that trees can increase soil infiltration rates, even in highly compacted soils such as those typically found in the urban environment. Only two studies quantified this increase, with Bartens et al. (2008) showing that tree roots increased soil infiltration rates by an average of 63% over unplanted controls and 153% for severely compacted soils. This same study demonstrated that trees can also increase infiltration rates in structural soils, with green ash grown in CU Soil having an infiltration rate 27 times greater than the unplanted CU Soil control sites (Bartens et al 2008). Kays (1980) showed a 35% decrease in suburban forest infiltration rates with removal of the understory and leaf litter. Chen et al (2014) identified soil rehabilitation with compost to be an important practice for mitigating urban soil compaction and also found the presence of trees contributes to an increase in soil hydraulic conductivity.

In a study of a stormwater biofilter, Le Coustumer et al. (2012) found that hydraulic conductivity declined over time for both vegetated and unvegetated biofilters, except those planted with the tree *Melaleuca ericifolia*. Hydraulic conductivity for the biofilter planted with *M. ericifolia* initially decreased from 155 to 100 mm/h over the first 40 weeks, but then increased to 295 mm/h after 60 weeks, finishing at around 240 mm/h at the end of testing (72 weeks). The authors hypothesize this is due to the importance of thick roots that help to maintain permeability of the soil over time through the creation of macropores.

Three other studies were reviewed that quantify the impact of trees on infiltration rates in non-urban environments. Mlambo et al. (2005) found that soil infiltration rates under tree canopy (0.12 +/- 0.02 mm/s) were 50% higher than outside the canopy (0.06 +/- 0.03 mm/s), and that infiltration rates were significantly higher under large trees than medium or small trees. Lal (1996) found that after the deforestation of a Nigerian forest, infiltration rates decreased by 20 to 30 percent. Wondzell and King (2003) summarized the literature on infiltration rates in burned and unburned forests of the Pacific Northwest and Rocky Mountain regions and showed that infiltration rates were around 35% lower in burned forests than unburned ones (value estimated from chart).

Runoff Reduction

The combined effect of trees' ability to intercept and evapotranspire rainfall and promote infiltration of water into the soil is that the rate and overall proportion of rainfall that becomes runoff is reduced. Most studies on runoff reduction provided by urban forests use hydrologic models to estimate the impact of trees on reducing stormwater runoff. The most commonly used models are American Forest's CITYgreen software, which is based on TR-55 (USDA SCS, 1986) and uses runoff curve numbers that predict runoff based on land use type, and the US Forest Service's i-tree (formerly known as UFORE), which is based on hydrodynamic canopy models. These modeling studies show that, as forest cover in a municipality or watershed increases, runoff decreases (and the inverse is also true).

Table 7 summarizes the results from the studies reviewed on runoff reduction by urban trees and forests. As indicated in the description in Table 7, each study has a unique approach to quantifying runoff reduction.

Table 7. Studies of Runoff Reduction by Urban Trees

Study	Results	Description
American Forests (1999)	19% increase in runoff	Modeled increase in runoff associated with loss of 14% forest cover
Armson et al. (2013)	58% reduction in runoff in summer and 62% in winter	Measured reduction from plot containing a tree pit and surrounded by asphalt
Wang et al. (2008)	2.6% runoff reduction	Modeled reduction associated with increasing tree cover over turf from 12 to 40%
	3.4% runoff reduction	Modeled reduction associated with increasing tree cover over impervious surface from 5 to 40%
Xiao and McPherson (2011b)	88.8% runoff reduction	Measured runoff reduction for bioswale integrating structural soils and trees[1]

Table 7. Studies of Runoff Reduction by Urban Trees

Study	Results	Description
Page et al. (2014)	80% runoff reduction	Measured runoff volume captured and treated by Silva Cell with tree[1]
Sanders (1986)	7% increase in runoff	Modeled increase in runoff associated with loss of 22% forest cover during a single storm
	5% reduction in runoff	Modeled reduction associated with increasing tree cover over non-surfaced areas from 37% to 50% during a single storm

[1] study did not include unplanted controls

Watershed-scale studies of runoff reduction often provide results in terms of the percent of annual runoff reduced by a given percent of tree cover in the watershed (in comparison to the runoff generated if trees were not present). These results can be translated into a percent runoff reduction per unit area of canopy if watershed areas are provided in the studies. However, not all studies are conducted on an annual basis and the results (streamflow measured at the watershed outlet) reflect not just the effect of trees in the watershed but the cumulative effect of all other land cover types and watershed features.

For both the CITYgreen and i-Tree models, analyses identical to those described in Table 6 have been conducted for dozens of municipalities across the US. Only the Chesapeake Bay region CITYgreen study was reviewed for this synthesis because the methodology is the same in all studies and this paper provides results that are most relevant to the Bay. The runoff curve number method upon which CITYgreen is based was developed for agricultural watersheds and has been shown to be relatively inaccurate in estimating runoff from forest (Tedela et a. 2012). Wang et al. (2008), Armson et al. (2013) and Herrera Environmental Consultants (2008) all found that runoff reduction was more pronounced when trees were planted over/near impervious cover. This is likely attributable to the greater amount of stormwater runoff generated on impervious surfaces, compared to turfgrass or other pervious areas.

In addition to reducing total runoff volume, tree canopy can delay peak runoff because of its ability to intercept and slowly release rainfall (Asadian and Weiler 2009). Research on the ability of tree canopy to delay throughfall reports a delay in throughfall of 0.17 hours to 3.7 hours after rainfall (Asadian 2010, Xiao 2000).

Studies of natural forests infer runoff reduction by measuring changes in runoff from streams draining forested basins before and after clearcutting. Table 8 summarizes these studies, which show a reduction in annual water yield of 8% to 80% after forest harvesting. In the Hornbeck et al. (1997) study, increases in annual water yield diminished rapidly as forests regenerated and were undetectable within 7-9 years after treatment. Douglas and Swank (1972) summarized 23 experiments from mixed deciduous hardwood forests in the Appalachian Highlands. They found a linear relationship between streamflow increase during the first year after forest removal and the percentage reduction of the forest stand, where first year increase = -1.43 + 0.13(% basal area reduction). Bosch and Hewlett (1982) conducted a review of 94 catchment experiments across the world as an update to a review by Hibbert (1967). Pine and eucalypt forest types were found to cause on average 40 mm change in water yield per 10% change in forest cover and deciduous hardwood and scrub ~25 and 10 mm, respectively.

Table 8. Runoff Reduction by Natural Forests

Study	Location	Results	Description
Hornbeck et al. (1997)	Hubbard Brook, New Hampshire	Annual water yields increased by an average of 32% after forest clearing in forested watersheds.	Measured by comparing streamflow in forested basins before and after deforestation.
Moore and Wondzell 2005	Oregon Cascades, Oregon Coast and South Coastal British Columbia	Mean changes in annual water yields after forest harvesting ranged from 8-43% in the Oregon Cascades, 14-26% in the Oregon Coast and South Coastal British Columbia and 15-80% in snow dominated small catchments.	Measured by comparing streamflow in forested basins before and after deforestation.

Water Quality

The primary way that urban trees affect water quality is by reducing the amount of stormwater runoff that reaches surface waters. Trees also improve soil and water quality through uptake of soil nutrients by plants and soil microbes. Tree roots stabilize the soil and tree canopies reduce the impact of raindrops, both of which reduce soil erosion. Trees, specifically the leaf litter produced by trees, are also a needed source of nutrients and carbon to support stream ecology and are deposited to stream from adjacent riparian areas, or through delivery of leaf litter from the urban drainage system. As discussed below, the research to quantify the excessive nutrient load delivered by leaf litter in urban watersheds is incomplete. Most of the studies reviewed focused on the effects of urban trees on the quality of stormwater runoff.

Effects of Trees on the Quality of Stormwater Runoff

Only one study directly addresses the effects of urban trees on the quality of stormwater runoff. Nine of the studies reviewed were field studies of the pollutant removal performance of stormwater treatment systems that include trees (e.g., Silva cells). However, only four of these studies (Denman 2006, Denman et al. 2011, Denman et. al 2015, Read et al. 2008) included unplanted controls to separate out the benefits provided by the tree vs. the filter media, and only one of those (Denman 2006) reported results that represent the water quality performance associated with the trees. Read et al. (2008) did not report results for trees versus other types of vegetation. In addition, the studies, which are summarized in Table 9, evaluate different species of nutrients and/or use varying methods to calculate percent pollutant removal.

Table 9. Pollutant Removal by Stormwater Treatment Systems with Trees *(note: the pollutant reductions are for the practice and not the effect of trees in the practice).*

Study	Treatment System Type	Parameter and % Reduction						
		TN	NOx	DIN	TKN	TP	FRP	TSS
Denman 2006	Street Tree Bioretention	82-95						
Denman et. al 2011; Denman et al. 2015	Biofiltration		2-78				70-96	
Geronimo et al. 2014	Tree Box Filter							80-98
Page et al 2014	Silva Cell				71, 84	72		86
Roseen et al. 2009	Street Tree			62		-54		88
UNHSC, 2012	Tree Box Filter (Non-proprietary)	10		8				88
UNHSC, 2012	Filterra	15				52		85
Xiao and McPherson 2011a	Bioswale	95.3[1]						95.5[2]

[1]average of all nutrient species results
[2]average of results from TSS and TDS

The values shown in Table 9 represent the percent removal of each pollutant provided by stormwater treatment systems with trees. Note that even where studies incorporated unplanted controls, the results reflect the pollutant removal of the entire system. Only the Denman (2006) study provides sufficient data to separate out the pollutant removal associated with just the trees. For the aforementioned study, the results show 82%, 85% and 95% removal of TN by the three bioretention systems with trees, compared to of -7%, 0%, and 36% removal by their respective unplanted controls. The difference between pollutant removal effectiveness of these planted and unplanted systems can be assumed to represent the enhanced TN reductions provided by the trees, with values of 59%, 85% and 89%.

Of the other studies on water quality benefits of urban trees, a modeling study by Band et al. (2010) estimated that current tree cover in Baltimore County, MD's Baisman Run watershed reduced TSS by 981 lbs (445kg) over the simulation period, TP by 4.4 lbs (2kg), TKN by 26.5 lbs (12kg) and NO2+NO3 by 8.8 lbs (4 kg). These results were based on modeling using UFORE-Hydro that simulated changes in flow due to changes in watershed land cover, and applied national median EMC values to estimate associated changes in pollutant loads; yet, it is difficult to put these values into context because the total pollutant inputs to the watershed are unknown. Matteo et al. (2006) ran a watershed-scale model of the water quality impacts of roadside and riparian buffers, but did not provide enough information about the area of the forested buffers to scale the results down to an individual tree planting site or forest plot. This is similar to the results presented by Goetz et al. (2003) and by the CITYgreen and i-Tree studies reviewed in the previous section in that the results are only applicable if the urban tree canopy credit is based on a percent tree canopy for a given watershed or municipality.

Groffman et al. (2009) measured nitrate leaching from urban forest and grasslands and found that annual nitrate leaching was higher in grass than in forest plots, except for one highly disturbed site that had hydrologic N losses well in excess of atmospheric inputs. Nitrate losses from forest plots in this study were 0.05 to 0.79 g N/m/yr; however, nitrogen inputs to the system were not measured. Another study by Groffman et al. (2004) found nitrate yields of 0.11 to 0.14 kg N/ha/yr and TN yields of 0.48 to 0.58 kg N/ha/yr from a forested basin, and estimated N retention of 95% by this basin, compared to 75% for a suburban basin and 77% for an agricultural basin.

Two studies were reviewed that address urban trees and water quality but do not specifically deal with stormwater runoff. Zhang et al. (2011) measured organochlorine pesticides in rainfall, canopy throughfall and runoff and found that the canopy was able to intercept 40% of the wet and dry deposited pollutants compared to a site with no trees, but further research is needed to determine the ultimate fate of the pollutants. Conversely, Xiao and McPherson (2011a) found that nutrients were added as rainfall passed through the tree canopy due to canopy leaching of pollutants that were previously deposited from atmospheric sources. The washoff of atmospheric deposition from leaves would not be considered an additional source of nutrients; however, tree canopy may delay the delivery of these nutrients to the stream.

Numerous studies have evaluated the water quality benefits of natural forests. Table 10 summarizes measured nutrient and sediment exports from undisturbed forests. It also presents ratios of pollutant loading from forests that have undergone disturbance (e.g., ice damage, insect defoliation, fire) and forests that were harvested (using a range of methods such as cattle grazing, clearcutting, strip cutting, and whole tree removal) compared to the pre-disturbance or control sites for those particular studies. Given the limited amount of data on the water quality benefits of urban trees and forests, the data from undisturbed forests could be applied to establish upper bounds of pollutant removal. The ratios for disturbed and harvested forest could potentially be useful if culled to look only at studies that represent conditions commonly found in urban forest patches or planting sites (e.g., sparse cover, die-off from lack of watering, compacted soils).

Table 10. Nutrient and Sediment Loads from Non-Urban Forests[1]

Type of Forest	**Pollutant Export (lbs/acre/year)[1] (n)**		
	TN	TP	TSS
Undisturbed	2.14[3] (123)	0.16[2] (14)	41.9[2] (17)
	Ratio of Pollutant Export from Harvested/Disturbed Forest:Reference[4]		
Disturbed	3.09	2.04	2.04
Harvested	7.03	3.12	3.05

[1] based on studies of eastern forests compiled by Justin Hynicka from Maryland DNR for urban tree canopy land use recommendations

[2] median value

[3] calculated as the sum of median values for NO3 and TKN

[4] mean ratio of harvested or disturbed pollutant export to pollutant export from reference sites

Pollutant Uptake

Most studies on pollutant uptake by trees focus on nutrient uptake by trees in the riparian zone. These studies were not included in the literature review because the focus of this work is on the benefits of upland urban trees. That is, processes such as nitrate removal from shallow groundwater and denitrification hotspots along the soil water interface are more prominent processes in riparian areas compared to tree in the upland area (Johnson et al 2013). A few studies were available from the field of phytoremediation—the process of using plants to remove contamination from soil and water— which show trees' potential to remove pollutants through plant uptake, adsorption and microbial activity. Phytoremediation has mainly been applied to remove metals, pesticides, and organic compounds from soil and groundwater but could potentially be applied to nutrients in stormwater runoff. Tree species typically used for phytoremediation include willow, poplar (cottonwood hybrids), and mulberry, because they have deep root systems, fast growth, a high tolerance to moisture, and are able to control migration of pollutants by consuming large amounts of water (Metro, 2002; IRTC, 2001; Shaw and Schmidt, 2007). Once pollutants are taken up by plants, one or more activities may occur. Pollutants can be moved into the above-ground portions of the plants, accumulate in the root zone, be broken down through natural processes of plant growth, or be transformed into inert material and discharged through plant leaves or shoots. Biological uptake is seen as only a temporary removal process because the pollutants may be returned to the system when the plant dies, unless it is harvested.

Leaf Litter

An emerging topic in urban stormwater management is the effect of nutrients and carbon from leaf litter on urban streams. Leaf litter represents a major energy source (DOC) and source of nutrients to streams where water soluble compounds readily leach from the leaves within hours to days following immersion, with macro-invertebrates and bacteria decompose the leaf material in-stream. In urban-suburban areas, leaf litter collects in curbs and gutters that is flushed through the storm drain system, contributing nutrients to urban streams that are generally already impaired for excessive nutrients, or impaired biota.

While many urban areas have less than 40% tree canopy, leaf litter input to streams from riparian and upland areas does occur. This results in a large and steady supply of leaves to streams (aka the "gutter subsidy"). In a recent Scientific and Technical Advisory Committee workshop report (Sample et al 2015) and Nowak (2014) provided data for Baltimore, MD estimating an urban tree canopy biomass nutrient load of 28.8 lbs/ac/yr and 2.95 lbs/ac/yr of N and P, respectively if all the leaves were accounted for in the load. However, it is known trees sequester nitrogen prior to leaf fall in trunks, roots and branches leaving low amount of nitrogen in leaf litter. In an outfall netting study in Easton, MD, Stack et al (2013) found an average of 4.7 TN lb/ac/yr and 0.36 TP lb/ac/yr associated with leaf litter in catchments with 24% canopy cover. The difference between these loading rates is attributed in part to the aged leaf litter at the outfall and leaf litter reaching the streams compared to the total canopy used to estimate the biomass by Nowak (2014). Street sweeping studies have also quantified the potential impact of leaf litter on urban nutrient loadings. Baker et al (2014) and Berretta et al. (2011) found that organic matter comprised 10% of the load collected by street sweepers. Waschbusch (2003) also found a similar estimate from a street sweeping study and this contributed to 30% of the total phosphorus load. This 'gutter subsidy' was estimated by Baker et al (2014) to be 2 lbs - 6 lbs P/curb-mile in residential catchments with up to 20% tree canopy. Templer et al (2015) found that up to 52 ± 17% of residential litterfall carbon (C) and nitrogen is exported through yard waste removed from the City of Boston, which is equivalent to more than half of annual N outputs as gas loss (i.e. denitrification) or leaching. While, recent studies illustrate the available supply of leaf litter in urban areas, further research is needed to better quantify the fate, transport, and processing of leaf litter in urban watersheds and how to best account for this source as part of an urban nutrient mass balance. While the nutrients, specifically phosphorus is likely a source to urban streams, it is unknown how much of this soluble load is represented in current monitoring of urban outfalls.

Summary

Urban tree canopy has great potential for helping to meet nutrient and sediment load reductions for the Chesapeake Bay TMDL. However, trees are unlike most other urban BMPs, which have a defined drainage area and are engineered to capture and remove pollutants from stormwater runoff. While trees affect processing of nutrients from the soil, atmosphere and groundwater, their primary impact on water quality is attributed to the prevention of water pollution by reducing the amount of runoff generated from areas where tree canopy is present. In the absence of tree canopy, rain falling on urban surfaces such as parking lots, streets and lawns picks up various pollutants as it runs off the landscape. Therefore, the cumulative effect of tree canopy is to temporarily detain rainfall and gradually release it, regulating the flow (volume and peak) of stormwater runoff downstream and thereby preventing pollutants in rainfall and on urban surfaces from being transported to local waterways.

The ability of an urban tree to reduce runoff is determined by how much rainfall is intercepted and evaporated in the canopy or infiltrated into the soil.   The removal of soil water by trees through transpiration also affects runoff by increasing soil water storage potential, effectively lengthening the amount of time before rainfall becomes runoff. By preventing rain from becoming runoff, trees decrease the volume of runoff that is available to pick up sediment and nutrients from the urban landscape. This correlation between runoff and water quality is widely accepted and many stormwater runoff models—including i-tree HYDRO, the Simple Method (Schueler 1987) and the Runoff Reduction Method (Hirschman et al. 2008)—-calculate pollutant loads as a product of runoff volume and pollutant concentration. Trees provide additional water quality benefits through uptake of pollutants from the atmosphere, soil and groundwater, and may contribute nutrients to surface waters through leaf litter, but these components are more challenging to quantify given the available data and its variability. Since the literature on hydrologic benefits of urban trees is much more plentiful than studies of water quality benefits, a possible avenue to explore for a credit is to model the connection between runoff reduction and pollutant reduction.

While these processes and mechanisms for reducing runoff and pollutants are well known, the amount by which trees reduce runoff is highly variable, and by extension water quality as well. For example, interception alone is influenced by numerous factors, including the intensity, duration and frequency of rainfall; canopy architecture, leaf area, leaf angle distribution, leaf surface characteristics; and meteorological factors such as wind speed and vapor pressure deficits. Evapotranspiration is similarly influenced by a number of environmental and structural factors. Studies that quantify these processes offer results that are often site-specific or event-specific. All of these factors present a challenge with translating these results into water quality credits that reflect the "average" condition. Best professional judgement is needed to develop recommendations that reflect the best available science while accounting for this variability as well as the average operational condition of the entire watershed.

References

American Forests. 1999. *Regional ecosystem analysis Chesapeake Bay Region and the Baltimore-Washington Corridor: Calculating the value of nature*. American Forests. Washington, DC.

Armson, D. P. Stringer, A.R. Ennos. 2013. The effect of street trees and amenity grass on urban surface water runoff in Manchester, UK. *Urban Forestry & Urban Greening*, 12(3): 282-286.

Asadian, Y. 2010. *Rainfall interception in an urban environment*. Master of Science Thesis, the University of British Columbia. Vancouver. BC.

Asadian, Y. and M. Wieler. 2009. A new approach in measuring rainfall intercepted by urban trees in coastal British Columbia. *Water Quality Research J. Can*. 44: 16-25.

Baker, L.A. S.E. Hobbie, P. Kalinosky, R. Bintner, and C. Buyarski. 2014. Quantifying nutrient removal by street sweeping. *Final Project Report to the Minnesota Pollution Control Agency,* Sept. 2014.

Band, L., Nowak, D., Yang, Y., Endreny, T., and J. Wang. 2010. Modeling in the Chesapeake Bay Watershed: effects of trees on stream flow in the Chesapeake Bay. Report to Forest Service for Agreement No.07-CO-11242300-145.

Bartens, J, S. D. Day, J. R. Harris, Dove, J.E., and T. M. Wynn. 2008. Can urban tree roots improve infiltration through compacted subsoils for stormwater management? *Journal of Environmental Quality*, 37: 2048-2057.

Bartens, J., Day, S. D., Harris, J. R., Wynn, T. M., and J. E. Dove. 2009. Transpiration and root development of urban trees in structural soil stormwater reservoirs. *Environmental Management*, 44: 646-657.

Berland, A. and M. E. Hopton. 2014. Comparing street tree assemblages and associated stormwater benefits among communities in metropolitan Cincinnati, Ohio, USA. *Urban Forestry and Urban Greening*, 13: 734-741.

Berretta, C., S. Raje, and J. Sansalone. 2011. Quantifying nutrient loads associated with urban particulate matter and biogenic/litter recovery through current MS4 source control and maintenance practices. *Final report to Florida Stormwater Association Educational Foundation*, University of Florida. Gainesville, FL.

Boggs, J. L., and G. Sun. 2011. Urbanization alters watershed hydrology in the piedmont of North Carolina. *Ecohydrology*, 4: 256-264.

Bolton, K., and M. Greenway. 1999. Nutrient sinks in a constructed melaleuca wetland receiving secondary treated effluent. *Water Science and Technology*, 40(3): 341-347.

Bosch, J. M. and J. D. Hewlett. 1982. A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. *Journal of Hydrology*, 55: 3-23.

Center for Watershed Protection (CWP). 2014. *Impervious cover credit for individual trees.* Memo prepared for Montgomery County Department of Environmental Protection.

Cermak, J., Hruska, J., Martinkova, M., and A. Prax. 2000. Urban tree root systems and their survival near houses analyzed using ground penetrating radar and sap flow techniques. *Plant and Soil*, 219: 103-116.

Chen, Y., Zhang, Z., Li, Z., Tang, J., Caldwell, P., and W. Zhang. 2011. Biophysical control of whole tree transpiration under an urban environment in northern China. *Journal of Hydrology*, 402: 388-400.

Chen, Y., Day, S., Wick, A., and K. McGuire. 2014. Influence of urban land development and subsequent soil rehabilitation on soil aggregates, carbon, and hydraulic conductivity. *Science of the Total Environment*, 494-495: 329-336.

Cienciala, E. ; Kučera, J. ; Lindroth, A. ; Čermák, J. ; Grelle, A. ; Halldin, S. 1997. Canopy transpiration from a boreal forest in Sweden during a dry year. *Agricultural and Forest Meteorology*, 86(3): 157-167.

Crockford, R.H., and D.P. Richardson. 2000. Partitioning of rainfall into throughfall, stemflow and interception: effect of forest type, ground cover and climate. *Hydrological Processes*, 14: 2903-2920.

Deguchi, A., Hattori, S., and H. Park. 2006. The influence of seasonal changes in canopy structure on interception loss: Application of the revised Gash model. *Journal of Hydrology*, 318: 80-102.

Denman, E.C. 2006. Are street trees and their soils an effective stormwater treatment measure? The 7th National Street Tree Symposium.

Denman, E.C., May, P.B., and G. M. Moore. 2015. The potential role of urban forests in removing nutrients from stormwater. Journal of Environmental Quality. Special section on The Urban Forest and Ecosystem Services.

Denman, E.C., P.B. May, and G.M. Moore. 2011. The use of trees in urban stormwater management. This paper has been presented previously at the Urban Trees Research Conference, “Trees, people and the built environment”, 13 & 14 April 2011, Birmingham, UK and the ISA Annual Conference, 25 – 27 April 2011, Sydney.

Douglas, J. E., and W. T. Swank. 1972. Streamflow modification through management of eastern forests. USDA Forest Service Research Paper SE-94. USDA Forest Service, Southeastern Forest Experimental Station, Ashville NC.

Ford, C., Hubbard, R., and J. Vose. 2011. Quantifying structural and physiological controls on variation in canopy transpiration among planted pine and hardwood species in the southern Appalachians. *Ecohydrology*, 4(2): 183-195.

Geronimo, F. K. F., M.C. Maniquiz-Redillas, J.A.S. Tobio, and L.H. Kim, L H. 2014. Treatment of suspended solids and heavy metals from urban stormwater runoff by a tree box filter. *Water Science and Technology,* 69(12): 2460-7.

Goetz, S., Wright, R., Smith, A., Zinecker, E., and E. Schaub. 2003. IKONOS imagery for resource management: Tree cover, impervious surfaces and riparian buffer analyses in the mid-Atlantic region. *Remote Sensing of Environment*, 88: 195-208.

Green, S.R. 1993. Radiation balance, transpiration and photosynthesis of an isolated tree. Agricultural and Forest Meteorology, 64(3): 201-221.

Groffman, P.M., Law, N., Belt, K., Band, L.E., and G. Fisher. Nitrogen fluxes and retention in urban watershed ecosystems. *Ecosystems*, 7: 393-403.

Groffman, P. M., C.O. Williams, R. V. Pouyat, L.E. Band, and I.D. Yesilonis. 2009. Nitrate leaching and nitrous oxide flux in urban forests and grasslands. *Journal of Environmental Quality,* 38(5): 1848-60.

Guevara-Escobar, A.; E. González-Sosa, C. Véliz-Chávez, E. Ventura-Ramos, and M. Ramos-Salinas. 2007. Rainfall interception and distribution patterns of gross precipitation around an isolated Ficus benjamina tree in an urban area. *Journal of Hydrology*, 333: 532-541

Guidi, W., Piccioni, E., and E. Bonari. 2008. Evapotranspiration and crop coefficient of poplar and willow short-rotation coppice used as a vegetation filter. *Bioresource Technology*, 99: 4832-4840.

Heal, K.V., Stidson, R.T., Dickey, C.A., Cape, J.N., and M.R. Heal. 2004. New data for water losses from mature Sitka spruce plantations in temperate upland catchments. Hydrological Sciences Journal, 49(3): 477-493.

Herrera Environmental Consultants. 2008. *The Effects of Trees on Stormwater Runoff*. Prepared for Seattle Public Utilities. Herrera Environmental Consultants, Seattle, WA.

Hibbert, A.R. 1969. Water yield changes after converting a forested catchment to grass. *Water Resources Research*, 5 (3): 634-640.

Hirschman, D. Collins, K., and T. Schueler. 2008. The runoff reduction method: technical memorandum. Center for Watershed Protection & Chesapeake Stormwater Network, Ellicott City, MD. Pages 1-25.

Hornbeck, J. W., Martin, C. W., and C. Eager. 1997. Summary of water yield experiments at Hubbard Brook Experimental Forest, New Hampshire. *Canadian Journal of Forest Research*, 27(12): 2043-2052.

Inkilainen, Elina, M. R. McHale, G. B. Blank, A. L. James, and E. Nikinmaa. 2013. The role of the residential urban forest in regulating throughfall: A case study in Raleigh, North Carolina, USA. *Landscape and Urban Planning*, 119: 91-103.

Interstate Technology Regulatory Cooperation (ITRC). 2001. Technical/Regulatory Guidelines: Phytotechnology Technical and Regulatory Guidance Document. ITRC.

Johnson, S.R., M. R Burchell III, R. O Evans, D. L.Osmond and J. W. Gilliam. 2013. Riparian buffer located in an upload landscape position does not enhance nitrate-nitrogen removal. *Ecological Engineering*, 52:252-261.

Jones, J.A. 2000. Hydrologic processes and peak discharge response to forest removal, regrowth, and roads in 10 small experimental basins, western Cascades, Oregon. *Water Resources Research.* 36(9): 2621-2642.

Kays, B.L. 1980. Relationship of forest destruction and soil disturbance to increased flooding in suburban North Carolina piedmont. Proceedings of the Third Conference of the Metropolitan tree Improvement Alliance, 118-125. The Metropolitan Tree Improvement Alliance.

Kestrel Design Group Team (KDGT). 2013. The Roles and Effects of Tree Evapotranspiration and Canopy Interception in Stormwater Management Systems and Strategies. MPCA Stormwater Manual Revisions. Prepared by the Kestrel Design Group Team for the Minnesota Pollution Control Agency.

Kirnbauer, M.C., B.W. Baetz, W.A. Kenney. 2013. Estimating the stormwater attenuation benefits derived from planting four monoculture species of deciduous trees on vacant and underutilized urban land parcels. *Urban Forestry and Urban Greening*, 12(3): 401-407.

Kjelgren, R. and T. Montague. 1998. Urban Tree Transpiration over Turf and Asphalt Surfaces. *Atmospheric Environment*. 32(1): 35-41.

Lal, R. 1996. Deforestation and land use effects on soil degradation and rehabilitation in western Nigeria I. soil physical and hydrological properties. *Land Degradation and Development*, 7: 19-45.

Le Coustumer, S., Fletcher, T. D., Deletic, A., Barruad, S., and P. Poelsma. 2012. The influence of design parameters on clogging of stormwater biofilters: a large-scale column study. *Water Research*, 46: 6743-6752.

Link, T.E., Unsworth, M., and D. Marks. 2004. The dynamics of rainfall interception by a seasonal temperate rainforest. *Agricultural and Forest Meteorology*, 124: 171-191.

Litvak, E., Bijoor, N. S., and D. E. Pataki. 2014. Adding trees to irrigated turfgrass lawns may be a water-saving measure in semi-arid environments. Ecohydrology, 7: 1314-1330.

Livesley, S.J., B. Baudinette and D. Glover. 2014. Rainfall interception and stem flow by eucalypt street trees – The impacts of canopy density and bark type. *Urban Forestry and Urban Greening,* 13(1): 192-197.

Matteo, M. T. Randhir and D. Bloniarz. 2006. Watershed-Scale Impacts of Forest Buffers on Water Quality and Runoff in Urbanizing Environment. *Journal of Water Resources Planning and Management,* 132(2): 144-15.

McPherson, G.E. and J. R. Simpson. 2002. A comparison of municipal forest benefits and costs in Modesto and Santa Monica, California, USA. *Urban Forestry & Urban Greening*, 11: 61–74.

McPherson, G., J. R. Simpson, Q. Xiao, and C. Wu. 2011. Million Trees Los Angeles canopy cover and benefit assessment. *Landscape and Urban Planning*, 99: 40-50.

Metro. 2002. Green streets: Innovative solutions for stormwater and stream crossings. Portland, OR. 144 p.

Mlambo, D., Nyathi, P., and I. Mapaure. 2005. Influence of *Colophospermum mopane* on surface soil properties and understorey vegetation in a southern African savanna. *Forest Ecology and Management*, 212: 394-404.

Molchanov, A. A. 1960. Gidrologicheskayo rol lesa (Hydrological role of forest). Izd-vo AN SSSR, Moscow, 488 pp.

Moore, R.D., and S.M. Wondzell. 2005. Physical hydrology and the effects of forest hydrology in the Pacific Northwest: a review. *Journal of the American Water Resources Association*, 41(4): 763-784.

Nowak, D. 2014. Personal communication. Urban tree canopy analysis for Baltimore City. As cited in Sample et al. 2015.

Page, J.L., R.J. Winston, and W.F. Hunt, III. 2014. Soils beneath pavements: An opportunity for stormwater control and treatment. *Ecological Engineering*. In press
Read, J., T. Wevill, T. Fletcher and A. Deletic. 2008. Variation among plant species in pollutant removal from stormwater in biofiltration systems. *Water Research*, 42(4-5): 893-902.

Pataki, D. E., Carreiro, M. M., Cherrier, J., Grulke, N. E., Jennings, V., Pincetl, S., Pouyat, R. V., Whitlow, T. H., and W. C. Zipperer. 2011. Coupling biogeochemical cycles in urban environments: ecosystem services, green solutions, and misconceptions. *Frontiers in Ecology and the Environment*, 9 (1): 27-36.

Peters, E. B., McFadden, J. P., and R. A. Montgomery. 2010. Biological and environmental controls on tree transpiration in a suburban landscape. *Journal of Geophysical Research*, 115(4).

Post, D.A., and J.A. Jones. 2001. Hydrologic regimes of forested, mountainous, headwater basins in New Hampshire, North Carolina, Oregon, and Puerto Rico. *Advances in Water Resources*, 24: 1195-1210.

Read, J., Wevill, T., Fletcher, T., and A. Deletic. 2008. Variation among plant species in pollutant removal from stormwater in biofiltration systems. *Water Research*, 42: 893-902.

Reynolds, E.R.C., F.B. Thompson, and United Nations University. 1988. Forests, Climate, and Hydrology: Regional Impacts. United Nations University, Tokyo, Japan. http://www.nzdl.org/fast-cgi-bin/library?e=d-00000-00---off-0unu--00-0--0-10-0---0---0prompt-10---4-------0-1l--11-en-50---20-about---00-0-1-00-0-0-11-1-0utfZz-8-00&cl=CL2.1&d=HASH4845310c3238b42b2d2574&gt=2.

Roseen, R., T.P. Ballestero, J.J. Houle, P. Avellaneda , J. Briggs, G. Fowler, and R. Wildey. 2009. Seasonal performance variations for storm-water management systems in cold climate conditions. *Journal of Environmental Engineering*. 135:128-137.

Sample, D., K. Berger, P. Claggett, J. Tribo, N. Goulet, B. Stack, S. Claggett, and T. Schueler. 2015. The peculiarities of pervious cover: A research synthesis on allocating pollutant loads to urban land uses in the Chesapeake Bay. STAC Publication Number 15-001, Edgewater, MD. 57 pp.

Sanders, R. 1986. Urban vegetation impacts on the hydrology of Dayton, Ohio. *Urban Ecology*, 9: 361-376

Schueler, T.R., 1987. Controlling urban runoff: a practical manual for planning and designing urban BMPs. Publication No. 87703. Metropolitan Washington Council of Governments, Washington, DC.

Shaw, D. and Schmidt, R. 2003. Plants for stormwater design. Species selection for the Upper Midwest. St. Paul, MN: Minnesota Pollution Control Agency. 369 p.

Sinclair, T.R., Holbrook, N.M., and N. Zwieniecki. 2005. Daily transpiration rates of woody species on drying soil. *Tree Physiology*, 25: 1469-1472.

Soares, A.L., Rego, F.C., McPherson, E.G., Simpson, J.R., Peper, P.J., and Q. Xiao. 2011. Benefits and costs of street trees in Lisbon, Portugal. *Urban Forestry & Urban Greening*, 10: 69-78.

Stack, B., N. Law, S. Drescher and B. Wolinski. 2013. Gross Solids characterization study in the Tred Avon Watershed Talbot County, MD. Prepared for Talbot County Department of Public Works. Center for Watershed Protection. Ellicott City, MD.

Tedela, N.H., McCutcheon, S.C., Rasmussen, T.C., Hawkins, R.H., Swank, W.T., Campbell, J.L., Adams, M.B., Jackson, C.R., and E.W. Tollner. 2012. Runoff curve numbers for 10 small forested watersheds in the mountains of the eastern United States. *Journal of Hydrologic Engineering* 17: 1188-1198.

Templer, P. H., Toll, J. W., Hutyra, L. R., and S. M. Raciti. 2015. Nitrogen and carbon export from urban areas through removal and export of litterfall. *Environmental Pollution*, 197: 256-261.

United States Department of Agriculture Soil Conservation Service (USDA SCS). 1986. Urban hydrology for small watersheds. Technical Release 55 (TR-55).

University of New Hampshire Stormwater Center. 2012. University of New Hampshire Stormwater Center 2012 Biennial Report.

Urban, J. 1999. Room to grow. *Treelink* 11:1-4.

Wang, J., T.A. Endreny, and D. J.Nowak. 2008. Mechanistic Simulation of Tree Effects in an Urban Water Balance Model. *Journal of the American Water Resources Association*, 44(1): 75-85.

Wang, H., Ouyang, Z., Chen, W., Wang, X., Zheng, H., and Y. Ren. 2011. Water, heat, and airborne pollutants effects on transpiration of urban trees. *Environmental Pollution*, 159: 2127-2137.

Wang, H. 2012. Transpiration rates of urban trees, *Aesculus chinensis*. *Journal of Environmental Sciences*, 24(7): 1278-1287.

Waschbush, R.J. 2003. Data and Methods of a 1999-2000 Street Sweeping Study on an Urban Freeway in Milwaukee County, Wisconsin. Open File Report 03-93. U.S. Department of the Interior, U.S. Geological Survey.

WQGIT (Chesapeake Bay Program Water Quality Goal Implementation Team). 2015. Protocol for the Development, Review, and Approval of Loading and Effetiveness Estimates for Nutrient and Sediment Controls in the Chesapeake Bay Watershed Model.

Wondzell, S., and J. King. 2003. Postfire erosional processes in the Pacific Northwest and Rocky Mountain regions. *Forest Ecology and Management*, 178: 75-87.

Wullschleger, S. D., Wilson, K. B., and P. J. Hanson. 2000. Environmental control of whole-plant transpiration, canopy conductance, and estimates of the decoupling coefficient for large red maple trees. *Agricultural and Forest Meteorology*, 104: 157-168.

Wullschleger, S. D., Hanson, P. J., and D. E. Todd. 200. Transpiration from a multi-species deciduous forest as estimated by xylem sap flow techniques. *Forest Ecology and Management*, 143: 205-213

Xiao, Q., E.G. McPherson, J.R. Simpson, and S.L. Ustin. 1998. Rainfall Interception by Sacramento's Urban Forest. *Journal of Arboriculture,* 24(4): 235-244.

Xiao, Qingfu, E.G. McPherson, S.L. Ustin, M.E. Grismer. 2000. A new approach to modeling tree rainfall interception. *Journal of Geophysical Research*, 105(D23): 29, 173-29,188.

Xiao, Q and E.G. McPherson. 2002. Rainfall Interception by Santa Monica’s Municipal Urban Forest. *Urban Ecosystems*, 6: 291-302.

Xiao, Q. and E.G. McPherson. 2011a. Performance of engineered soil and trees in a parking lot bioswale. *Urban Water Journal,* 8(4): 241-253.

Xiao, Q. and E.G. McPherson. 2011b. Rainfall interception of three trees in Oakland, California. *Urban Ecosystems*, 14: 755-769.

Yang, J.L., G.L. Zhang. 2011. Water infiltration in urban soils and its effects on the quantity and quality of runoff. *J Soils Sediments*.11: 751-761.

Zhang, W., Y. Youbin; Y. Tong, L. Ou, D. Hu, and X. Wang. 2011. Contribution and loading estimation of organochlorine pesticides from rain and canopy throughfall to runoff in an urban environment. J*ournal of Hazardous Materials*, 185(2-3): 801-806.

Annotated Bibliography

American Forests, 1999 (Runoff reduction)

Using CityGreen software, forest loss from 1973-1999 was calculated for a 1.5 million acre portion of the Chesapeake Bay region near the Baltimore-Washington corridor. During the study time period, average tree cover went from 51% to 37% and areas with heavy tree cover declined from 55% to 37%. Tree loss resulted in a 19% increase in runoff (for each 2 year peak storm event), an estimated 540 million ft^3 of water. In the study area, the existing tree canopy reduces the need for retention storage by 540 million cubic feet. The model relies on modified formulas from TR-55 to estimate stormwater runoff.

Armson et al. 2013 (Runoff reduction)

This study assessed the impact of trees upon urban surface water runoff by measuring the runoff from $9m^2$ plots covered by 1) grass, 2) asphalt, and 3) asphalt with a tree planted in the center. It was found that, while grass almost totally eliminated surface runoff, the tree plots significantly reduced runoff, with 26% runoff in winter and 20% in summer (as a percentage of rainfall). The trees and their associated tree pits reduced runoff from asphalt by 58% in the summer and 62% in winter. The reduction was attributed primarily to infiltration into the tree pit and canopy interception, although the tree's canopy covered about 35% of the plot. Relative to its canopy crown, the runoff reduction by the tree was estimated to be 170% in summer and 145% in winter.

Bartens et al. 2008 ((Infiltration)

This study examined whether tree roots can penetrate compacted subsoils and increase infiltration rates in the context of an infiltration BMP that uses structural soils and includes large canopy trees. One goal of the study was to determine if tree roots would grow into the compacted subsoils typically found under/adjacent to such a practice. The study found that tree roots increased soil infiltration rates by an average of 63%, and as much as 153%, over unplanted controls.

Bartens et al. 2009 (ET)

In this study, two trees were grown in structural soil mixes and were subject to three simulated infiltration rates for two growing seasons. Reduced infiltration rates were correlated with lower transpiration rates. Transpiration rates for one growing season were reported to be 0.80 to 1.14 $\mu g/cm^2/s$ for the green ash (depending on soil treatment) and 0.76 to 1.39 $\mu g/cm^2/s$ for the swamp white oak. The study also found that larger trees can take up more total water than smaller trees with higher transpiration rates.

Berland and Hopton, 2014

This study estimated canopy interception by street trees along geographic and demographic gradients in Cincinnati. Using i-tree, interception ranged from 59.2 to 214.3 m3 per km of effective street length. The mean interception value used in the model was 6.7m3 per tree, which the researchers note may overestimate runoff reduction.

CWP, 2014

Data from i-tree STREETS was used to plot the volume of rainfall intercepted per year versus trunk diameter and the trunk diameter versus age of the tree. Polynomial regressions were generated from these plots. Regression functions all had R^2 values of at least 0.999. The functions were tied and plotted for 3 tree species found in Montgomery County, MD and for the average "Broadleaf Deciduous Large" value from the i-Tree database for the Piedmont south climate region. I-tree uses a computer model described in Xiao et al. (1998) to generate rainfall interception. The statistical analysis showed an average annual interception volume of 2,000 gallons per tree for a 15-20 year old tree that is 9-15" DBH.

Denman 2006

Study of the performance of a pilot scale street tree bioretention system in reducing nitrogen loads in urban stormwater. Three tree species were planted in three soils of different hydraulic conductivity and irrigated with synthetic stormwater, along with 3 unplanted soil profiles used as controls and irrigated with tap water. The trees grew well in the irrigated soil. Nitrogen content (ammonium, oxidized nitrogen and organic nitrogen) of leach water was measured. Leached nitrogen loads were significantly reduced in systems with a tree. Compared to the total nitrogen input, the load leached in December 2004 from the L. confertus profiles following a 5 hour collection period was 95% less for the low SHC, 85% for the medium SHC and 82% for the high SHC soils. In the unplanted profiles the low SHC soil reduced nitrogen by 36%, whereas the medium (0%) and high SHC soils (-7%) did not remove nitrogen. This study does not appear to be peer reviewed.

Denman et al. 2011

Similar study design as above but this study measured soluble N and P in leachate. Some seasonal variability was found, with higher leaching of N and P in the warmer months. Again, tree growth was good. No significant differences in evergreen versus the one deciduous species planted. P removal did not occur until after the first summer. This study showed greater variability than the previous one. The NOx reduction provided by soils with trees, averaged over time, ranged from 2% to 78%. Reduction of filterable reactive phosphorus ranged from 70% to 96%. No specific values were provided for the unplanted controls for comparison. This study does not appear to be peer reviewed.

Geronimo et al. 2014

This study evaluated pollutant removal and runoff reduction by a tree box filter. The system reduced runoff by 40% for a hydraulic loading rate of 1m/day. It was found out that the hydraulic loading rate was dependent on the total runoff volume received by the system. TSS removal ranged from 80% to 98% at varying hydraulic loading rates. No unplanted control site was tested to evaluate the effects of the tree versus other mechanisms; however the study states that the filtration capacity of the tree box filter was presumed to be the main pollutant removal mechanism.

Groffman et al. 2009

This study measured nitrate (NO3) leaching and soil:atmosphere nitrous oxide (N2O) flux in four urban grassland and eight forested long-term study plots with a range of disturbance, soil type and landscape position in the Baltimore, Maryland metropolitan area from 2002-2005. Annual NO3 leaching ranged from 0.05 to 0.79 g N m yr for the forest plots and was lower than in grass plots, except in a very dry year and when a disturbed forest plot was included in the analysis. Although NO3 leaching was higher in urban grasslands than in forest plots, the difference was not as large or consistent as expected, and the most intensively fertilized plots did not have the highest leaching losses. The N2O results were even more surprising because there were few differences between forest and grass plots, and, again, the more intensively fertilized grasslands did not have greater fluxes. These results suggest that N cycling in urban grasslands is complex and that there is significant potential for N retention in these ecosystems. Grass plots consistently produced less leachate volume than forest plots. It is suspected that the difference was due to higher evapotranspiration on the grass plots due to higher soil temperatures and the longer growing season in urban grassland versus forest ecosystems. A complication in the leaching comparisons was the fact that one of our forest plots was extensively disturbed and had very high N losses. Although leaching from most of the forest plots was very low, consistent with many previous studies of forest ecosystems, data from our highly disturbed forest plot showed that forests can have hydrologic N losses well in excess of atmospheric inputs. Likely causes of the high N losses from the highly disturbed forest plot include soil disturbance and invasion by exotic plant and earthworm species. These results suggest that not all forest components of urban landscapes are functioning as strong N sinks.

Guevara-Escobar et al 2007

This work evaluated rainfall interception and distribution patterns of gross precipitation around the canopy of a single evergreen tree Ficus benjamina (L.) in Queretaro City, Mexico. Nineteen individual storms occurring from July to October, 2005, were analyzed. Interception loss was 59.5% of gross rainfall and was primarily attributed to evaporation, which was not limited due to the low relative humidity and high temperatures. The study showed a screen effect of the tree crown on gross precipitation and if not accounted for in study designs, will lead to underestimation of interception losses. The screen effect was important and accounted for 18.7% of the interception losses by the tree canopy alone.

Herrera Environmental Consultants 2008

This report reviews the literature on the effects of trees on stormwater runoff and make recommendations for applying the available research to develop a stormwater credit for urban trees in the City of Seattle. The review found that evergreen trees in the Pacific Northwest can intercept on average 20% of annual rainfall (18-25%, depending on season) and can transpire 10% of precipitation. Modeling two scenarios of an evergreen tree planted over 1)an impervious surface and 2) a lawn, and based on the value identified above, the authors estimate that planting a tree over impervious cover results in a 27% reduction in the amount of rainfall that becomes runoff (95% runoff coefficient assumed for impervious cover) and planting a tree over turf results in a 12% reduction in the amount of rainfall that becomes runoff (20% for turf). The result for tree planted near impervious cover approach 30%, a value also suggested in the literature on runoff reduction. The same exercise was repeated for deciduous trees using values of 10% for interception and 5% for transpiration. The authors recommend a credit of 30% of the canopy footprint for evergreens and 15% for deciduous trees, if the tree is located within 10 feet of an impervious surface. Trees located more than 10 feet from an impervious surface would receive half this credit.

Inkiläinen et al 2013

To quantify the amount of rainfall interception by vegetation in a residential urban forest this study measured throughfall in Raleigh, NC, USA between July and November 2010. Throughfall comprised 78.1–88.9% of gross precipitation, indicating 9.1–21.4% rainfall interception. Cumulative rainfall interception over the study period ranged from 9.1- 10.6 and the storm based values ranged from 19.9-21.4. Canopy cover and coniferous trees were the most influential vegetation variables explaining throughfall whereas variables such as leaf area index were not found significant in our models. The results do not appear to reflect interception by trees but are for the entire residential parcel which includes other land cover types.

Kays 1980

Infiltration tests conducted across a North Carolina watershed on various land use types found that a medium aged pine-mixed hardwood forest had a mean final constant infiltration rate of 31.56 inches per hour. When the forest understory and leaf litter were removed, the resultant lawn had a mean infiltration rate of 11.20 inches per hour.

Kirnbauer et al. 2013

i-Tree Hydro was used to derive a simplified Microsoft Excel-based water balance model to quantify the canopy interception potential and evaporation for four monoculture planting schemes on urban vacant lots, based on 7 years (2002–2008) of historical hourly rainfall and mean temperature data in Hamilton, Ontario, Canada. The results demonstrate that the tree canopy layer was able to intercept and evaporate approximately 6.5%–11% of the total rainfall that falls onto the crown across the 7 years studied, for the G. biloba, P.×acerifolia and A. saccharinum tree stands and 17%–27% for the L. styraciflua tree stand. This study revealed that the rate at which a species grows, the leaf area index of the species as it matures, and the total number of trees to be planted need to be determined to truly understand the behavior and potential benefits of different planting schemes.

Kjelgren and Montague 1998

The study used a two-layer canopy model to study transpiration of tree species as affected by energy-balance properties of a vegetated (turf) and paved surface. Trees over asphalt had consistently higher leaf temperature, than those over turf, apparently due to interception of the greater upwards long-wave radiation flux from higher asphalt surface temperatures. In one study flowering pear over asphalt in a humid environment had higher leaf temp resulting in one-third more total water loss compared to trees over turf. In other studies, however, water loss of green ash and Norway maple over asphalt in an arid environment was either equal to or less than that over turf. Less water loss was due to higher leaf temp over asphalt causing prolonged stomatal closure. Model manipulation indicated that tree water loss over asphalt will depend on the degree of stomatal closure resulting from how interception of increased energy-fluxes and ambient humidity affect leaf-to-air vapor pressure differences.

Livesley et al. 2014

This study measured canopy throughfall and stemflow under two eucalypt tree species in an urban street setting over a continuous five month period. The species with the greater plant area index intercepted more of the smaller rainfall events, such that 44% of annual rainfall was intercepted as compared to 29% for the less dense E. saligna canopy. Stemflow was less in amount and frequency for the roughbarked E. nicholii as compared to the smooth barked species. However, annual estimates of stemflow to the ground surface for even the smoothbarked E. saligna would only offset approximately 10mm of the 200mm intercepted by its canopy. This study provides an evidence base for tree canopy impacts upon urban catchment hydrology, and suggests that rainfall and runoff reductions of up to 20% are quite possible in impervious streetscapes.

Matteo et al. 2006

This study used the generalized watershed loading function model to evaluate watershed-wide impacts of best management practices (BMPs) scenarios representing riparian and street buffers on water quality, quantity, and open space in rural, suburban, and urbanized environments. The proportion of urban forest cover reduced sediment and nutrient loading, decreased stormwater runoff, and increased groundwater recharge in urbanizing watersheds. The model simulated runoff, groundwater recharge, ET, and TN and TP loads for 4 scenarios in each of the 3 settings: 1) baseline, 2) 10 foot roadside tree buffers, 3) 200 foot riparian buffers, and 4) both the riparian and roadside buffers. Results for the suburban catchment were: TSS reduction of 1.83% from baseline, TN 0.06% reduction, TP 2.75% reduction, runoff 5.24% reduction, ET increase of 0.06% and increase in groundwater recharge of 1.67%. Results for the urban catchment were: TSS reduction of 4.24% from baseline, TN 6.59% reduction, TP 6.57% reduction, runoff 8.75% reduction, ET increase of 2.74% and increase in groundwater recharge of 33.84%. However, the total area of forest associated with each scenario was not reported, making it difficult to apply the result to the individual tree planting site scale. There is also a question about the CNs used in the model for forest (46 for rural forest, 65 for suburban forest and 30 for urban forest), which were taken from TR-55 but the value used for urban forest is for A soils and woods in good condition, and produces less runoff than the suburban and rural sites.

McPherson and Simpson 2002

This paper presents a comparison of the structure, function, and value of street and park tree populations in two California cities. Modesto is covered by 31% trees, while Santa Monica has 15% tree cover. A numerical interception model accounted for the amount of annual rainfall intercepted by trees, as well as throughfall and stem flow (Xiao et al. 1998). The volume of water stored in tree crowns (m3/tree) was calculated from crown projection areas (area under tree dripline), leaf areas, and water depths on canopy surfaces. Hourly meteorological and rainfall data for 1995 (Modesto) and 1996 (Santa Monica) were used as input. Urban forests in Modesto were estimated to reduce stormwater runoff by 3.2m2 per tree, and by 7.0 m2/tree in Santa Monica. Interception differed between cities because of variables such as annual rainfall pattern and tree foliation periods.

McPherson et al 2011

The purpose of this study was to measure Los Angeles's existing tree canopy cover (TCC), determine if space exists for 1 million additional trees, and estimate future benefits from the planting using i-tree. A numerical interception model accounted for the amount of annual rainfall intercepted by trees, as well as throughfall and stem flow (Xiao et al. 1998). The volume of water stored in tree crowns (m3/tree) was calculated from crown projection areas (area under tree dripline), leaf areas, and water depths on canopy surfaces. Hourly meteorological and rainfall data for 1995 (Modesto) and 1996 (Santa Monica) were used as input. Over the 35-year span of the project, planting of 1 million trees was estimated to reduce runoff by approximately 51 to 80 million m3. The average annual interception rate per tree ranged from a low of 0.4m3 for the crapemyrtle (representative of small trees in the inland zone) to a high of 5.6m3 for the jacaranda (representative of medium trees in the inland zone). The difference is related to tree size and foliation period. The crapemyrtle is small at maturity and is deciduous during the rainy winter season, whereas the jacaranda develops a broad spreading crown and is in-leaf during the rainy season.

Page et al 2104

This study evaluated the hydrologic and water quality performance of two suspended pavement systems using Silva cells in North Carolina. Both were planted with a crepe myrtle but no controls were used to test the influence of the trees on results. Pollutant concentrations were significantly reduced, including TP, TN and TSS. TP reductions were at least 72% and TSS reductions were greater than 86%. TN results were not reported but TKN reductions were 71% and 84%. 80% of runoff at the inlet was captured and treated by the practices. Peak flow was mitigated by 62% for stormwater not generating bypass.

Read et al. 2008

Study authors used a pot trial of 20 Australian species to investigate how species vary in the removal of pollutants from semisynthetic stormwater passing through a soil filter medium. Unplanted controls were used that were irrigated with tap water. Five tree species were included in the mix. While plant species improved pollutant removal compared to unvegetated systems (especially for N and P), the study did not provide specific removal values for tree species versus non tree species.

Roseen et al. 2009

This study monitored pollutant removal performance of 6 LID systems from 2004-2006 to evaluate seasonal variations in performance and the influence of cold climates on performance. These were contrasted with data from conventional and manufactured systems. One of the systems was a street tree/filter. Parameters monitored included TSS, TP, dissolved inorganic N, total Zinc and total petroleum hydrocarbons- diesel range. Seasonal performance evaluations indicate that LID filtration designs differ minimally from summer to winter, while smaller systems dependent largely on particle settling time demonstrated a marked winter performance decline. Frozen filter media did not reduce performance. Reported results for the street tree: efficiency ratios of 88% for TSS, 62% for DIN, and -54% for TP. The efficiency ratio was determined to be a more stable estimate of pollutant removal than removal efficiency because it weighs all storms equally and reflects overall influent and effluent concentrations across the entire dataset.

Sivyer et al. 1997

This study used a pan evaporation model to develop a method for predicting irrigation amount and frequency for street trees and tested it on two newly planted deciduous tree species in Norfolk, VA. The calculated daily transpiration rate for a 3” caliper tree during the growing season was estimated at 2.7 gallons per day.

Soares et al. 2011

This study used i-tree to quantify the value of street trees in Lisbon, Portugal. A numerical interception model accounted for the amount of annual rainfall intercepted by trees, as well as through fall and stem flow. The model estimated that Lisbon’s street trees intercepted approximately 186,773m3 of rainfall annually. On average, each tree intercepted 4.5m3 annually. This estimate was considered to be conservative because the rainfall data used were from a year with lower than normal rainfall.

The Kestrel Design Group 2013

In this paper, literature on ET and rainfall interception are reviewed to provide a basis for quantifying these functions as they relate to stormwater BMPs in the State of Minnesota’s stormwater crediting calculator. The paper reviews the various methods for quantifying ET, including direct versus indirect measure approaches, hydrological, micrometeorological and plant physiology approaches, as well as analytical versus empirical approaches. The authors review the advantages and disadvantages of each approach and recommend use of the Lindsey-Bassuk single whole tree water use equation for estimating ET and crediting trees for associated reductions in runoff. The Lindsey-Bassuk equation requires canopy diameter, leaf area index, evaporation rate per unit of time and evaporation rate as inputs and sources of information for each input are identified.

Wang et al. 2008

This study used the UFORE model, which simulates hydrological processes of precipitation, interception, evaporation, infiltration, and runoff using data inputs of weather, elevation, and land cover along with nine channel, soil, and vegetation parameters. The model was tested in the urban Dead Run catchment of Baltimore, Maryland. Total predicted tree canopy interception was 18.4% of precipitation. Key findings included: trees significantly reduce runoff for low intensity and short duration precipitation events; as LAI increases, interception rate increases as well; trees over impervious cover have a greater runoff reduction effect than trees over turf; as potential evaporation increases, interception increases; greater relative interception was seen with lower intensity storms; increasing tree cover over turf from

12% to 40% resulted in 2.6% runoff reduction; and increasing tree cover over IC from 5% to 40% resulted in 3.4% runoff reduction.

Xiao and McPherson 2003

A mass and energy balance rainfall interception model was used to simulate rainfall interception processes for street and park trees in Santa Monica, CA. Annual rainfall interception by the 29,299 trees was 193,168 m3 (6.6 m3/tree), or 1.6% of total precipitation. Rainfall interception ranged from 15.3% (0.8 m3/tree) for a small *Jacaranda mimosifolia* (3.5 cm diameter at breast height) to 66.5% (20.8 m3/tree) for a mature *Tristania conferta* (38.1 cm). In a 25-year storm, interception by all street and park trees was 12,139.5 m3 (0.4%), or 0.4 m3/tree. Rainfall interception varied seasonally, averaging 14.8% during a 21.7 mm winter storm and 79.5% during a 20.3 mm summer storm for a large, deciduous *Platanus acerifolia* tree.

Xiao and McPherson 2011a

A rainfall interception study was conducted in Oakland, California to determine the partitioning of rainfall and the chemical composition of precipitation, throughfall, and stemflow. Rainfall interception measurements were conducted on a gingko (Ginkgo biloba) (13.5 m tall deciduous tree), sweet gum (Liquidambar styraciflua) (8.8 m tall deciduous), and lemon tree (Citrus limon) (2.9 m tall broadleaf evergreen). The lemon, ginkgo, and sweet gum intercepted 27.0%, 25.2% and 14.3% of gross precipitation, respectively. The lemon tree was most effective because it retained its foliage year-round, storing more winter rainfall than the leafless ginkgo and sweet gum trees. Stemflow was more important for the leafless sweet gum. Because of its excurrent growth habit and smooth bark, 4.1% of annual rainfall flowed to the ground as stemflow, compared to less than 2.1% for the lemon and 1.0% for the ginkgo.

Xiao and McPherson 2011b

A bioswale integrating structural soil and trees was installed in a parking lot to evaluate its ability to reduce storm runoff, pollutant loading, and support tree growth. The adjacent control and treatment sites each received runoff from eight parking spaces and were identical except the control used native soils. A tree was planted at both sites. Storm runoff, pollutant loading, and tree growth were measured. The bioswale reduced runoff by 88.8% and reduced solids (TSS, TDS) by 95.5% and minerals (TP, TKN, NH4, NO3)) by 95.3%. It appears the reductions were calculated based on comparison to that of a control. No runoff was generated at the treatment site for storm events less than 9 mm (70% of events). The engineered soil provided better aeration and drainage for tree growth than did the control’s compacted urban soil.

Xiao et al 1998

A one-dimensional mass and energy balance model was developed to simulate rainfall interception in Sacramento County, California. Annual interception was 6% and 13% of precipitation falling on the urban forest canopy for the City of Sacramento and suburbs, respectively. Summer interception at the urban forest canopy level was 36% for an urban forest stand dominated by large, broadleaf evergreens and conifers (leaf area index = 6.1) and 18% for a stand dominated by medium-sized conifers and broadleaf deciduous trees (leaf area index = 3.7). For 5 precipitation events with return frequencies ranging from 2 to 200 years, interception was greatest for small storms and least for large storms.

Xiao et al 2000

A rainfall interception measuring system was developed and tested for open-grown trees. The system was tested on a 9-year-old broadleaf deciduous tree (pear, Pyrus calleryana `Bradford') and an 8-year-old broadleaf evergreen tree (cork oak, Quercus suber) representing trees having divergent canopy distributions of foliage and stems. Interception losses accounted for about 15% of gross precipitation for the pear tree and 27% for the oak tree. Interception losses were attributed primarily to canopy storage. The results also showed that interception losses relative to rainfall decreased with increasing rainfall depth. The analysis of temporal patterns in interception indicates that it was greatest at the beginning of each rainfall event. Rainfall frequency is more significant than rainfall rate and duration in determining interception losses.

Yang and Zhang 2011

In this study the physical and chemical properties of urban soils were characterized for 30 urban sites representing a mix of land cover types and age of development. Three of the site types contained trees and were also the oldest sites (20-30 years) with the least amount of compaction (normal to light). Lawns with trees had the highest final infiltration rate, followed by trees with shrubs but the infiltration rate for these two categories was not significantly different. The highest final infiltration rate was comparable to that of a forest. Measured infiltration rate values for these two land cover types were not provided in the paper.

Zhang et al. 2011

This study was conducted to estimate the fluxes of organochlorine pesticides in rain and canopy throughfall and their contributions to runoff in Beijing. Runoff, rain and canopy throughfall sampling was conducted over a two year period at 3 sites, two of which were completely paved and one of which had a canopy area of 54m2 from landscaping trees. At the impervious sites, the contribution of hexachlorobenzene (HCB) and hexachlorocyclohexanes (HCH)s from rainfall accounted for approximately 50% of the mass in runoff. At the site with significant coverage of landscaping trees, the HCB, HCHs, and DDTs from the net canopy throughfall accounted for approximately 10% of the mass in the runoff. The pollutant concentrations in canopy throughfall represent a combination of wet deposition and the portion of dry deposition that is washed from the canopy during a storm. For some sampling dates, concentrations were higher in rainfall than throughfall, indicating that the leaves may have been relatively clean prior to the storm event and the canopy was therefore able to intercept the pollutants, at least temporarily. Further research is needed to evaluate the effects of retention capacity of leaves, antecedent dry days, and storm characteristics on pollutant concentrations in throughfall.

Appendix D: Tree Mortality Rate Literature Review

(database provided by J. Mawhorter)

Source	State	City	Annual Mortality Rate	Survey period	Sample Size	Tree type
McPherson, G. 2014.	CA	Los Angeles	0.044	2006 - 2010	98	Street
McPherson, G. 2014.	CA	Los Angeles	0.046	2006 - 2010	96	Yard
Roman, L., J. Battles, J. McBride. 2014.	CA	Sacramento	0.066	2007-2012	436	Yard
Ko, Y., J. Lee, G. McPherson, L. Roman. 2015.	CA	Sacramento	0.041	1991-1994	254	Yard
Ko, Y., J. Lee, G. McPherson, L. Roman. 2015.	CA	Sacramento	0.038	1994-2013	92	Yard
Roman, L., J. Battles, J. McBride. 2014.	CA	West Oakland	0.037	2007-2011	995	Street
Koeser, A., E. Gilman, M. Paz, C. Harchick. 2014.	Florida	NA	0.013	2005-2010	2354	NA
Jack-Scott, E., M. Piana, B. Troxel, C. Murphy-Dunning, M. Ashton. 2013.	Connecticut	New Haven	0.025	1995-2011	NA	Street
Jack-Scott, E., M. Piana, B. Troxel, C. Murphy-Dunning, M. Ashton. 2013.	Connecticut	New Haven	0.041	1995-2011	NA	Yard
Jack-Scott, E., M. Piana, B. Troxel, C. Murphy-Dunning, M. Ashton. 2013.	CT	New Haven	0.025	1995-2011	NA	Park
Jack-Scott, E., M. Piana, B. Troxel, C. Murphy-Dunning, M. Ashton. 2013.	CT	New Haven	0.027	1995-2011	NA	Vacant lot
Koeser, A., R. Hauer, K. Norris, R. Krouse. 2013.	WI	Milwaukee	0.011	1989-2005	793	Street
Koeser, A., R. Hauer, K. Norris, R. Krouse. 2013.	WI	Milwaukee	0.019	1979-1989	895	Street
Mincey, S. and J. Vogt. 2014.	IN	Indiana	0.017	2006-2012	6366	NA
Roman, L. and F. Scatena. 2011.	PA	Philadelphia	0.034	1995-2005	151	Street
Nowak, D., J. McBride, R. Beatty. 1990.	CA	Oakland	0.188	1985-1988	480	Street
Sullivan, M. 2004.	CA	San Francisco	0.029	5	1987	Street
Foster, R. and J. Blaine. 1978.	MA	Boston/Beacon Hill	0.061	10	215	Street
Foster, R. and J. Blaine. 1978.	MA	Boston/Bolyston St.	0.140	4	136	Street
Gates, S. and J. Lubar. 2007.	PA	Philadelphia	0.029	2	50	Street
Gates, S. and J. Lubar. 2007.	PA	Philadelphia	0.044	1.5	571	Street
Gates, S. and J. Lubar. 2007.	PA	Philadelphia	0.067	1	705	Street
Miller, R.H. and R.W. Miller. 1991.	WI	Milwaukee	0.124	6	1003	Street
Miller, R.H. and R.W. Miller. 1991.	WI	Waukesha	0.065	6	677	Street
Miller, R.H. and R.W. Miller. 1991.	WI	Stevens Point	0.070	9	368	Street

Lu, J., E. Svendsen, L. Campbell, J. Greenfeld, J. Braden, K. King, N. Raymond. 2010.	NY	New York	0.044	2	45094	Street
Lu, J., E. Svendsen, L. Campbell, J. Greenfeld, J. Braden, K. King, N. Raymond. 2010.	NY	New York	0.036	6	2417	Street
Lu, J., E. Svendsen, L. Campbell, J. Greenfeld, J. Braden, K. King, N. Raymond. 2010.	NY	New York	0.034	8	5053	Street
Lu, J., E. Svendsen, L. Campbell, J. Greenfeld, J. Braden, K. King, N. Raymond. 2010.	NY	New York	0.029	9	5935	Street
Sklar, F. and R. Ames. 1985.	CA	Oakland	0.175	1978-1980	1500	Street
Boyce, S. 2010.	NY	New York City/TriBeCa	0.016	2005-2009	503	Street
Thompson et al 2004	IA	NA	0.023	4	932	NA

Appendix E: Summary of i-Tree Forecast Results for Development of UTC BMP Credit Development

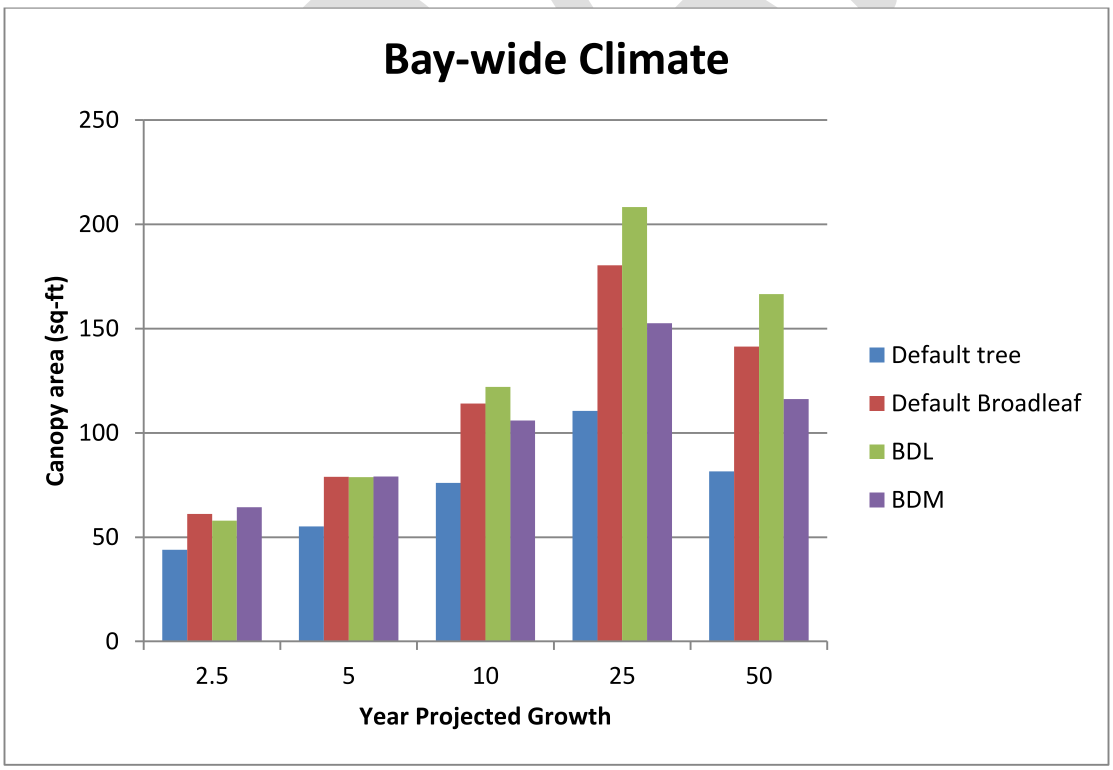
Tree canopy acreage (ft^2) for a single tree planted given a Bay-wide climate (166 FFD).

BDL = broadleaf large; BDM = broadleaf medium; CEL = coniferous evergreen large; CEM = coniferous evergreen medium

A1	Bay-wide (166 FFD)
Age	**Default tree**
2.5	44
5	55
10	76
25	111
50	82

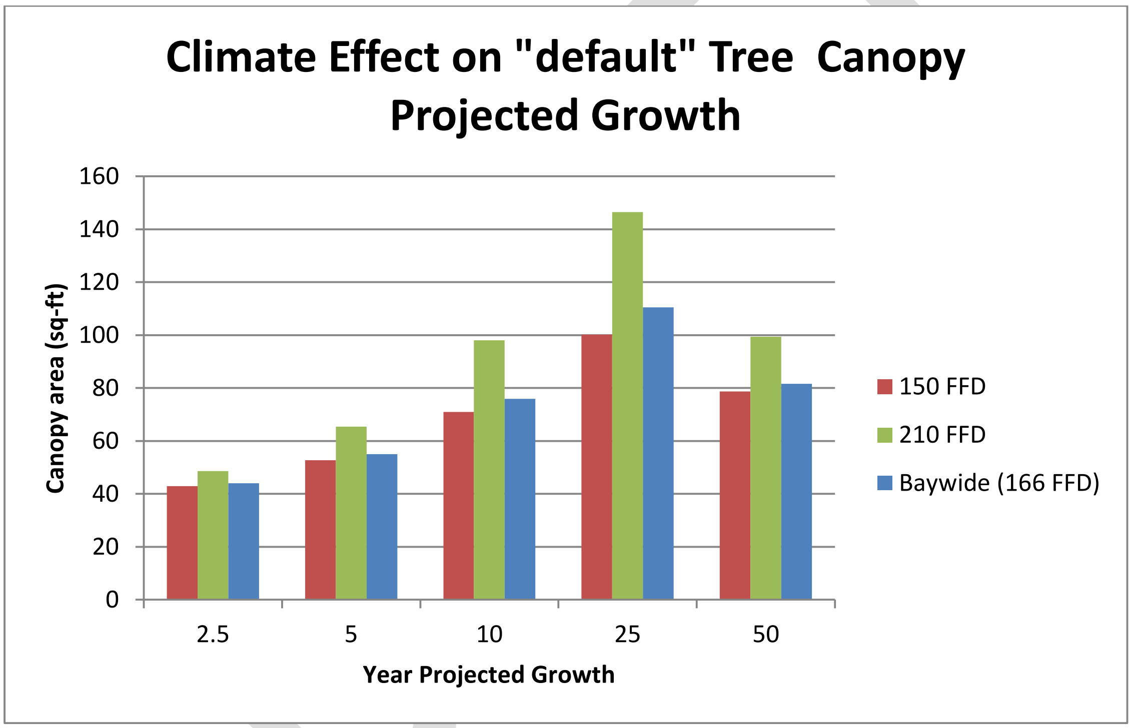
A2	Bay-wide	
Age	**BDL/BDM**	**CEL/CEM**
2.5	61	18
5	79	25
10	114	37
25	180	58
50	141	44

A3	Bay-wide	
Age	**BDL**	**BDM**
2.5	58	64
5	79	79
10	122	106
25	208	153
50	167	116



Tree canopy acreage (ft^2) for a single tree planted given two climate regions representative of the Bay Watershed (150 FFD and 210 FFD) for a 'default' tree

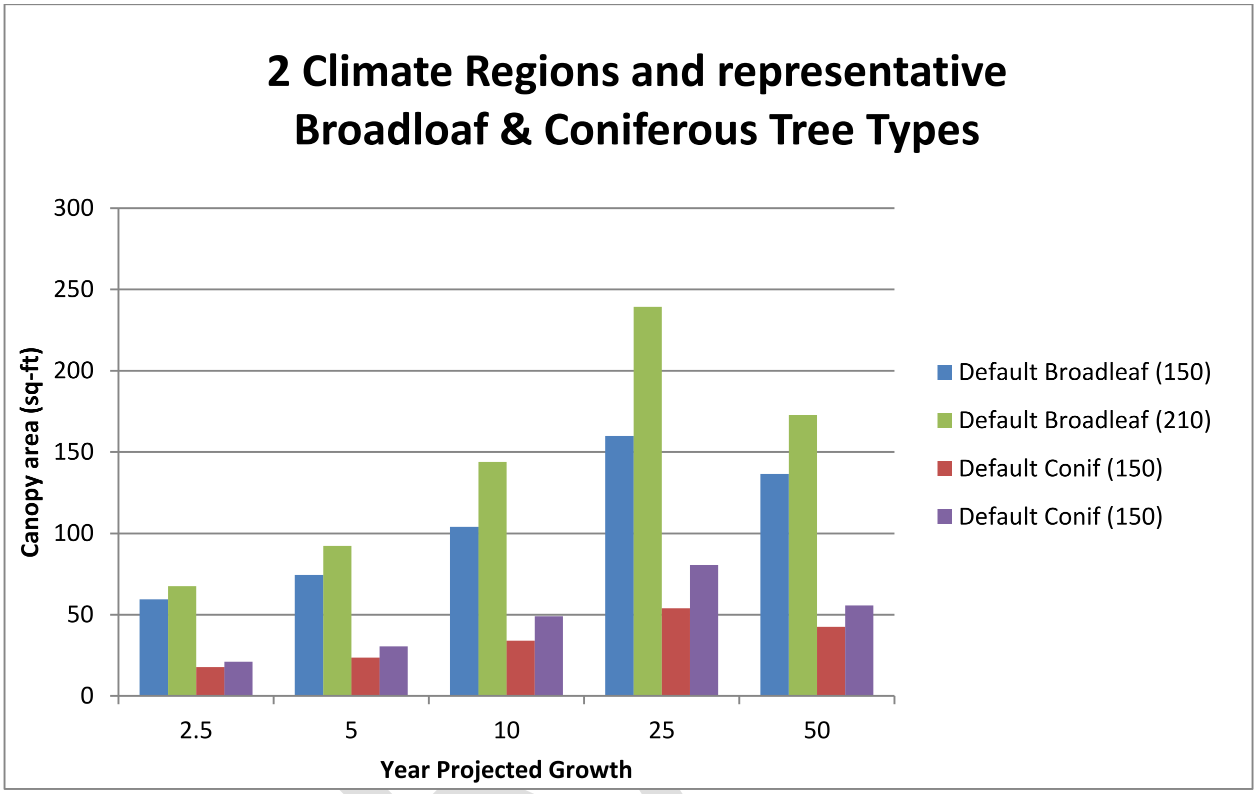
B1	150 FFD	210 FFD
Age	Default tree	Default tree
2.5	43	49
5	53	65
10	71	98
25	100	147
50	79	99



Tree canopy acreage (ft^2) for a single tree planted given two climate regions representative of the Bay Watershed (150 FFD and 210 FFD) for a representative broadleaf tree and coniferous tree. These are based on the average tree canopy of the large and medium tree species for both broadleaf and coniferous trees simulated in i-Tree Forecast.

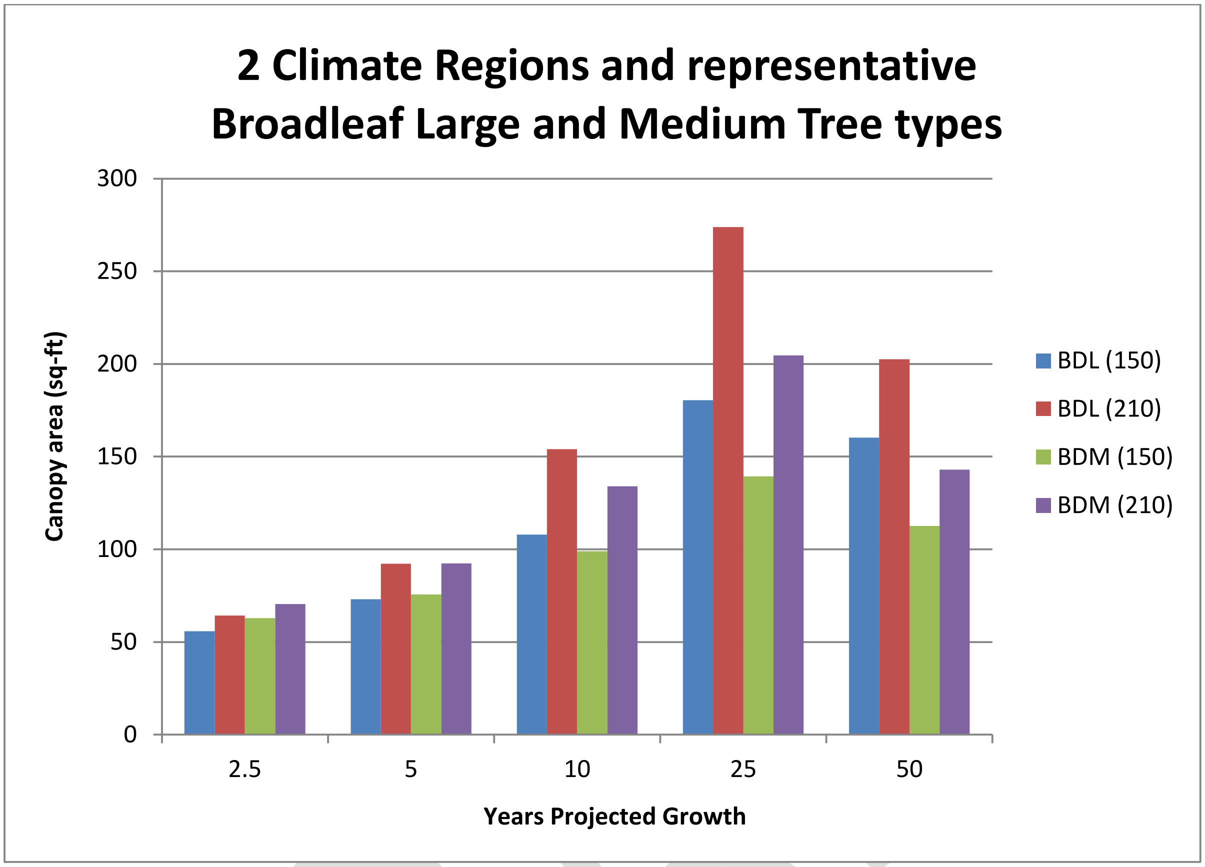
B2	150		210	
Age	BDL/BDM	CEL/CEM	BDL/BDM	CEL/CEM
2.5	59	18	67	21

5	74	24	92	31
10	104	34	144	49
25	160	54	239	80
50	136	42	173	56



Tree canopy acreage (ft²) for a single tree planted given two climate regions representative of the Bay Watershed (150 FFD and 210 FFD) for BDL and BDM tree types

B3	**150**		**210**	
Age	**BDL**	**BDM**	**BDL**	**BDM**
2.5	56	63	64	71
5	73	76	92	92
10	108	99	154	134
25	180	139	274	205
50	160	113	202	143


2 Climate Regions and representative
Broadleaf Large and Medium Tree types
Canopy area (sq-ft)
300
250
200
150
100
50
0
2.5
5
10
25
50
Years Projected Growth
BDL (150)
BDL (210)
BDM (150)
BDM (210)

Appendix F: Technical Requirements to Enter UTC BMP into Scenario Builder - DRAFT

Posted as a separate document for consideration by CBP partnership and Watershed Technical Workgroup. Will be included in full length report following WQGIT approval.

Appendix G: Conformity with BMP Review Protocol

The BMP review protocol established by the Water Quality Goal Implementation Team (WQGIT, 2015) outlines the expectations for the content of expert panel reports. This appendix references the specific sections within the report where the panel addressed the requested protocol criteria.

1. **Identity and expertise of panel members:** *See Table in Section 1*

2. **Practice name or title:** *Urban Tree Canopy*

3. **Detailed definition of the practice:** *See section 2 for detailed definition Urban Tree Canopy BMP*

4. **Recommended N, P and TSS loading or effectiveness estimates:** *See Ttable 8 in Section 5.2 and Appendix B*

5. **Justification of selected effectiveness estimates:** *See Appendix B*

6. **Description of how best professional judgement was used, if applicable to determine effectiveness estimates:** *See Appendix A, with additional information in Sections 5.2 and 5.3 related to the creditable area of this BMP*

7. **Land uses to which BMP is applied:** *Tree canopy over impervious and Tree Canopy of Pervious land uses in the future Phase 6 WSM.*

8. **Load sources that the BMP will address and potential interactions with other practices:** *TheUTC BMP will address pollutant laodings from Tree Canopy over Impervious and Tree Canopy over Pervious land uses in the Bay watershed. The report recommendations provide qualifying conditions to report the BMP as a BMP as part of new development, redevelopment and/or retrofit. Urban nutrient management may be applied to Tree Canopy Over Pervious Land uses.*

9. **Description of pre-BMP and post-BMP circumstances and individual practice baseline:** See Appendix B

10. **Conditions under which the BMP works/not works:** *Section2 describes conditions under which the BMP would not apply*

11. **Temporal performance of BMP including lag times between establishment and full functioning:** *No lag time is assumed.*

12. **Unit of measure:** *Acres*

13. **Locations in CB watershed where the practice applies:** *Urban*

14. **Useful life of the BMP:** *For the purposes of this report, the useful life of the practice is a minimum of 10 years but may be longer based on the Chesapeake Bay Program update of the Tree Canopy over Impervious and Tree Canopy over Impervious land uses*.

15. **Cumulative or annual practice:** Cumulative

16. **Recommended description of how BMP will be tracked and reported:** *See Section 6 for discussion of how state governments can track and report to the Bay Program.*

17. **Guidance on BMP Verification:** *See Section 6*

18. **Description of how the practice may be used to relocate pollutants to a different location.** ***See Appendix B***

19. **Suggestion for review timeline; when will additional information be available that may warrant re-evaluation of the practice effectiveness estimates:** *2025, or sooner pending outcome of research needs identified in Section 7.*

20. **Outstanding issues:** *See Section 7 for a discussion of future research needs to address outstanding issues related to the UTC BMP*

21. **Documentation of dissenting opinion(s) if consensus cannot be reached:** *not applicable*

22. **Operation and Maintenance requirements and how neglect alters the practice effectiveness estimates.** *The method used to derive the creditable area for this BMP accounts for mortality that would result if proper maintenance is not provided. Therefore, not other alterations in practice effectiveness is provided.*

23. A brief summary of BMP implementation and maintenance costs estimates, when this data is available through current literature: ***Not applicable***

24. **Technical Appendix:** *See Appendix F*